Habitat Restoration Plan for the Middle Rio Grande

Prepared for:

MIDDLE RIO GRANDE
ENDANGERED SPECIES ACT COLLABORATIVE PROGRAM
HABITAT RESTORATION SUBCOMMITTEE

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Report Qualification

The Middle Rio Grande ESA Collaborative Program (Program) is composed of federal, state, tribal, and local governments and nongovernmental entities with interests in finding solutions that contribute to the survival and recovery of the silvery minnow and flycatcher, while protecting existing New Mexico water rights and obligations under the Rio Grande Compact.

The New Mexico Interstate Stream Commission, as part of its commitment to the Program, retained Tetra Tech EM, Inc. to develop this plan with input from the Program's Habitat Restoration Subcommittee. The intent of the plan is to present a framework for the Program to implement and integrate actions needed to address both water and endangered species management issues in the Middle Rio Grande. Many of the Habitat Restoration Subcommittee participants and reviewers are representatives of state, federal, and municipal entities and non-governmental organizations. However, the contents of this publication do not necessarily reflect the views and policies of these entities and may not fully represent the opinions of individual participants.

Pueblo and tribal lands and resources occur within the Program area but are not included in activities under the Program without the express written consent of these entities. This document has not been formally reviewed by tribal entities and it is not intended to represent the policies and views of the Middle Rio Grande Pueblos.

Front Cover: Middle Rio Grande at Los Lunas (Mike Marcus); Rio Grande Silvery Minnow (U.S. Fish and Wildlife Service); Southwestern Willow Flycatcher (Tucson Citizen)

Back Cover: Portion of the first official soil map of the Middle Rio Grande valley below Albuquerque, circa 1912. Map highlights groundwater depths, overflow areas, gravel bars and islands. U.S. Dept. of Agriculture, Bureau of Soils and New Mexico Agricultural Experiment Station. 1914. Soil Survey of the Middle Rio Grande Valley Area, New Mexico. Government Printing Office, Washington, D.C.

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1.0 INTRODUCTION

The Rio Grande stream system, including its tributary aquifers, provides the water supply for the majority of New Mexico's citizens and is fundamental to the economics of the region. Water management activities were implemented throughout the 20th century to address needs related to flood protection, consistent water supplies, conveyance depletions, and water delivery obligations for New Mexico under the Rio Grande Compact (Compact). The highly variable and limited water supply of this region necessitated these actions as New Mexico's population expanded. As New Mexico enters the 21st century, the need for water management has increased to accommodate broader environmental considerations.

The various water management activities that fostered agricultural and urban developments along the Rio Grande also produced detrimental impacts on the habitats of species now listed as Federal and State endangered species (i.e., *listed species*). In particular, the Rio Grande silvery minnow (silvery minnow; *Hybognathus amarus*) and southwestern willow flycatcher (flycatcher; *Empidonax traillii extimus*) are Federal and State endangered species with habitat in the Middle Rio Grande (MRG) basin. The purposes of the Middle Rio Grande Endangered Species Act Collaborative Program (Program), as defined in its Draft Program Document dated April 21, 2003, "are to protect and improve the status of listed species in the Middle Rio Grande while simultaneously protecting existing and future water uses and to contribute to the recovery of those species, all the while complying with state and federal law, including Compact delivery obligations."

The Program's Habitat Restoration Subcommittee (Subcommittee) was given the responsibility for developing a comprehensive habitat restoration plan for the Program. The intent of the following sections is to present a framework plan to implement and integrate actions needed to address both water and endangered species management issues in the MRG. The Subcommittee intends to develop reach-specific habitat restoration plans over the next two years. These plans will evaluate current habitat conditions in greater detail, define opportunities for improvement, and establish priorities for habitat restoration sites and/or activities along the defined priority reaches.

Tetra Tech EM Inc. (Tetra Tech) was retained by the New Mexico Interstate Stream Commission (NMISC), as part of its commitment to the Program, to aid the Subcommittee in developing this Habitat Restoration Plan. After Tetra Tech prepared the first draft of this plan, Subcommittee participants provided review comments. Section 4 of the draft was discussed at length through a facilitated process to completely consider restoration techniques, purposes, and their effects. A draft of Section 4, including information specifically related to environmental impact analysis, was delivered to the Program's National Environmental Policy Act/Endangered Species Act (NEPA/ESA) Subcommittee. Finally, Tetra Tech prepared additional iterations of the plan, leading to this version, in an effort to capture the views of the Subcommittee participants. Many Subcommittee participants and reviewers are representatives of New Mexico State, Federal, and municipal entities and non-governmental organizations. However, the contents of this publication do not necessarily reflect the views and policies of these entities and may not fully represent the opinions of individual participants.

1.1 MRG Endangered Species Act Collaborative Program

The Program is composed of Federal, State, Tribal, and local governments and non-governmental entities with interests in finding solutions that contribute to the survival and recovery of the silvery minnow and flycatcher, while respecting existing New Mexico water rights and obligations under the Compact. Detailed descriptions of the goals and structure of the Program are presented in its Draft Program Document, which is substantively complete and will be finalized pending the results of the NEPA process. The Program was established to address issues associated with consultation under Section 7 and 10 of the ESA.

The area included in the Program (i.e., Program area or MRG) is defined in the Draft Program Document as encompassing the headwaters of the Rio Chama watershed and the Rio Grande, including tributaries, from the New Mexico-Colorado state line downstream to the location upstream of Elephant Butte Reservoir equaling the elevation at the Elephant Butte Dam spillway crest (4,450 feet above mean sea level [msl], Fig. 1-1). Pueblo and tribal lands and resources within the Program area are not included in activities under the Program without the express written consent of these entities. As indicated below, the Program area differs from the area designated as critical habitat for the silvery minnow.

1.2 Critical Habitat and Constraints on Habitat Restoration

Under the ESA, critical habitat is the specific geographical area occupied by a threatened and endangered species (TES) "on which are found those physical and biological features (I) essential to the conservation of the species and (II) which may require special management considerations or protection...." Section 3(5)(C) places constraints on the area that may be included as critical habitat, stating that, "(e)xcept in those circumstances determined by the Secretary, critical habitat shall not include the entire geographical area which can be occupied by the threatened or endangered species."

In 2003, the U.S. Fish and Wildlife (Service) defined critical habitat for the silvery minnow in the MRG (68 FR 8088). The designated area extends from Cochiti Dam downstream about 157 mi (252 km) to the utility line crossing the Rio Grande in Socorro County; this crossing occurs upstream of Elephant Butte Reservoir at the same elevation as the downstream spillway crest of the dam. The lateral limits (width) of critical habitat extend between the existing levees or, in areas without levees, 300 feet (91.4 m) into the riparian zone adjacent to each side of the bankfull stage of the Rio Grande. Portions of Santo Domingo, Santa Ana, Sandia, and Isleta Pueblo lands fall within the broader area designated as critical habitat but are specifically excluded from the definition. Furthermore, developed lands that lack primary constituent elements and are not essential to the recovery of the silvery minnow are excluded from the critical habitat designation. According to the Service (FWS, 2003a), these features include developed flood control facilities, existing paved roads, bridges, parking lots, dikes, levees, diversion structures, railroad tracks, railroad trestles, water diversion and irrigation canals outside of natural stream channels, the low flow conveyance channel (LFCC), active gravel pits, cultivated agricultural land, and residential, commercial, and industrial developments.

The Service has not defined critical habitat for the flycatcher in the MRG, or elsewhere. When the Service listed the flycatcher as endangered, a decision was deferred regarding the 643 mi (1,038 km) of riparian habitat proposed in the draft listing as critical habitat (FWS, 1995). The Service determined that it was necessary to consider additional comments, reconsider the prudence of designating critical habitat, and reconsider the boundaries of the critical habitat. A second period for public comment was opened in 1995. After considering the additional comments and scientific information received, the Service finalized the critical habitat designation identifying 599 mi (964 km) of riparian habitat (FWS, 1997a; FWS 1997b). During 2001, the 10th Circuit Court of Appeals set aside this critical habitat designation and instructed the Service to issue a new critical habitat designation in compliance with the Court's ruling. The Service is in the process of re-proposing critical habitat for the flycatcher. The final designation is expected in June 2004, unless otherwise instructed by the Court.

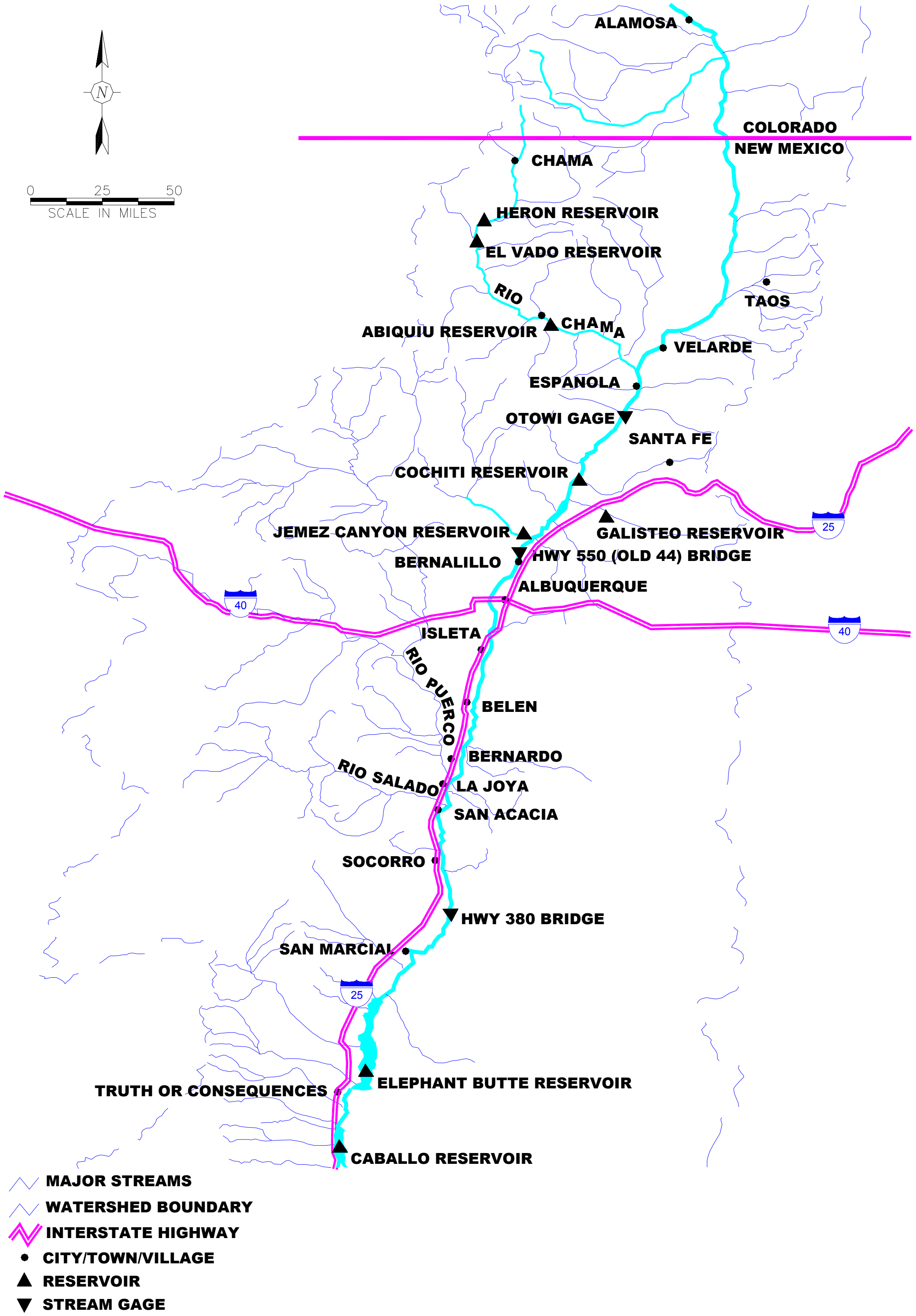

Figure 1-1: Middle Rio Grande Watershed

The Program identified various legal and institutional constraints that must be considered and/or addressed in association with its activities, including habitat restoration:

- Limitation of total depletions of native Rio Grande waters upstream of Elephant Butte Dam in New Mexico to the New Mexico apportionment defined by the Compact
- Consumptive use of imported San Juan-Chama Project waters as provided by the Colorado River and Upper Colorado River Basin Compacts for its authorized, contracted, and legal purposes, as provided by contracts and in accordance with the San Juan-Chama Project authorizing legislation
- The Colorado River, Upper Colorado River, and Rio Grande interstate compacts
- State water laws (permits, priority administration, beneficial use, etc.) and valid state water rights
- Pueblo and tribal water rights
- Federal trust responsibilities to affected Pueblos and Tribes
- Existing authorities and appropriations of all Federal and non-Federal entities
- Federal environmental compliance laws, including the ESA and NEPA
- All applicable State, Tribal, and Federal laws and regulations
- Applicable court decisions
- State directives or decisions
- State, tribal, and federal water quality laws, standards, and regulations
- Existing and future water users/uses
- Existing infrastructure
- Delivery obligations of the Middle Rio Grande Conservancy District (MRGCD)

In addition, Program activities are constrained by the level of funding allocated by Congress and the ability of non-Federal signatories to provide matching funds as required by the Congressional allocations.

1.3 2003 Biological Assessment and Biological Opinion

On March 17, 2003, the Service issued a Biological Opinion (BO) on the effects of actions associated with the "Programmatic Biological Assessment of Bureau of Reclamation's Water and River Maintenance Operations, Army Corps of Engineers' Flood Control Operation, and Related Non-Federal Actions on the Middle Rio Grande, New Mexico" (FWS, 2003a). The consultation involves two Federal agencies, the U.S. Bureau of Reclamation (Reclamation) and the U.S. Army Corps of Engineers (Corps), and two non-Federal entities, the NMISC and the MRGCD. The Service concluded that water operations and river maintenance activities in the Middle Rio Grande, as proposed in the February 19, 2003 Biological Assessment (BA), are likely to jeopardize the continued existence of the silvery minnow and flycatcher and adversely modify critical habitat of the silvery minnow (FWS, 2003a).

The BO presents numerous Reasonable and Prudent Alternatives (RPAs) to avoid the likelihood of jeopardizing the continued existence of the silvery minnow and flycatcher and of destruction or adverse modification of silvery minnow habitat (FWS, 2003a). These RPAs address issues of flow, habitat maintenance and restoration, captive propagation and augmentation, and water quality. Nine RPAs in the BO define "Specific Habitat Improvement Elements," as presented with their element letters used in the BO:

> P) Action agencies, in coordination with parties to the consultation, shall prevent or minimize destruction of potential or suitable flycatcher habitat when installing pumps or groundwater wells and coordinate with the Service prior to their installation if this action may affect flycatcher habitat.

Q) Action agencies, in coordination with parties to the consultation, shall improve gaging and real-time monitoring of water operations to provide dependable, accurate readings, including installation of gages near Los Lunas, and Highway 380, and all diversions, drains, returns and main ditches.

R) Reclamation, in coordination with the Service and parties to the consultation, shall complete fish passage at San Acacia Diversion Dam to allow upstream movement of silvery minnows by 2008. Reclamation and parties to the consultation, in coordination with the Service and Isleta Pueblo, shall work to complete fish passage at Isleta Diversion Dam, located on lands owned by Isleta Pueblo, by 2013.

S) In consultation with the Service and appropriate Pueblos and in coordination with parties to the consultation, action agencies shall conduct habitat/ecosystem restoration projects in the Middle Rio Grande to increase backwaters and oxbows, widen the river channel, and/or lower river banks to produce shallow water habitats, overbank flooding, and regenerating stands of willows and cottonwood to benefit the silvery minnow, the flycatcher, or their habitats. Projects should be examined for depletions. It is the Service's understanding that the objective of the action agencies and parties to the consultation is to develop projects that are depletion neutral. By 2013, additional restoration totaling 1,600 acres (648 hectares) will be completed in the action area. In the short term (5 years or less), the emphasis for silvery minnow habitat restoration projects shall be placed on river reaches north of the San Acacia Diversion Dam. This restoration will be distributed throughout the action area. Habitat restoration projects fulfilling RPA element J, from the June 29, 2001, biological opinion, shall be completed. The action agencies and parties to the consultation, in coordination with the Service, shall develop timetables and prioritize areas for restoration. Projects should result in the restoration/creation of blocks of habitat 24 hectares (60 acres) or larger. Consultation with the Service for each site will tier to this biological opinion.

T) When bioengineering (as described in Reclamation's biological assessment) cannot be used in Reclamation river maintenance projects, habitat restoration will be implemented to offset adverse environmental impacts resulting from river alteration. Habitat restoration efforts should replace the ecological functions and values of the affected area, both temporally and spatially.

U) Action agencies, in coordination with parties to the consultation, shall collaborate on the river realignment and proposed relocation of the San Marcial Railroad Bridge project, which is necessary to increase the safe channel capacity within the Middle Rio Grande. Construction for the relocation of the San Marcial Railroad Bridge will be initiated by September 30, 2008.

V) Each year that the NRCS [Natural Resource Conservation Service] April 1 Streamflow Forecast is at or above average at Otowi and flows are legally and physically available, the Corps shall bypass or release floodwater during the spring to provide for overbank flooding. The overbank flooding will be used to create an increased number of backwater habitats for the silvery minnow and flycatcher. The timing, amount, and locations of overbank flooding will be planned each year in conjunction with the Service and may be conducted in coordination with compact deliveries.

W) The Corps, in coordination with the Pueblo of Santa Ana, shall investigate and increase sediment transport through Jemez Canyon Dam. The Corps, in coordination with the Pueblo of Santo Domingo, shall also investigate and increase sediment transport through Galisteo Dam. By December 31, 2007, the Corps, in coordination with Cochiti Pueblo, shall complete an environmental baseline study and investigate the feasibility of transporting sediment from Cochiti Lake. The environmental baseline study shall address the issue of contaminated sediment raised by Cochiti Pueblo in comments received in response to the draft biological opinion. Prior to the release of any sediment from Cochiti Lake, the Corps shall conduct government-to-government consultations with Cochiti Pueblo as well as other downstream Pueblos that may be affected by this action. The action agencies and parties to the consultation shall investigate other locations in which sediment transport could be improved.

X) Action agencies, in coordination with parties to the consultation and in consultation with the Service, shall prevent encroachment of saltcedar on the existing channel and destabilize islands, point bars, banks, or sand bars in the Angostura, Isleta, and San Acacia Reaches. The methods used and areas proposed for destabilization should be agreed upon by the Service, Reclamation, the Corps, and appropriate Pueblos and landowners. This activity should not adversely affect flycatcher habitat. This action should be undertaken where reaches are dry and the Service encourages the action agencies and parties to the consultation to begin this action during the summer of 2003. Projects should be examined for depletions. It is the Service's understanding that the objective of the action agencies and parties to the consultation is to develop projects that are depletion neutral.

In addition, the BO includes two Water Quality Elements, with water quality being recognized as an important component of aquatic habitat (FWS, 2003a):

DD) With the increased emphasis and importance of the Angostura Reach for silvery minnow conservation, it is imperative that the addition of treated wastewater to the river provides water quality conditions protective of silvery minnow. The protective concentration of total residual chlorine (chlorine) for silvery minnow is less than or equal to 0.013 mg/L. The protective concentration of ammonia, as nitrogen [ammonia] (at 25 °C and pH 8), for silvery minnow is less than or equal to 3.09 mg/L for larvae and less than or equal to 9.3 mg/L for post-larvae.

EE) Action agencies, in coordination with parties to the consultation, shall provide funding for a comprehensive water quality assessment and monitoring program in the Middle Rio Grande to assess water quality impacts on the silvery minnow. This assessment and monitoring program should use available data from all sources.

Additional RPAs associated with water operations are presented and discussed in Section 2.4.5

1.4 Habitat Restoration Subcommittee Planning Objectives

The mission of the Subcommittee is to develop and lead implementation of a plan to restore habitat and restore river function that benefits the recovery of listed species and the ecosystem upon which they depend, within the legal and institutional constraints of the Program. This plan will serve as a guide for implementing habitat restoration activities for the Program by providing a framework for the solicitation, review, and implementation of habitat restoration proposals that will create long-term, self-sustaining habitat for the silvery minnow and flycatcher. This habitat must be consistent with the contemporary and attainable hydrologic and sediment regime of the Rio Grande. This plan is a technical resource for the development and assessment of the restoration activities conducted under the auspices of the Program. Specific objectives of the Subcommittee that will be addressed through this plan, and its subsequent reach-specific plans, include:

- Characterize the physical, biological, and ecological requirements for successful restoration of silvery minnow and flycatcher habitats in the MRG.
- Develop methods for restoring and improving silvery minnow and flycatcher habitats.
- Identify restoration opportunities that have the highest likelihood of benefiting the listed species from both an immediate and long-term perspective.
- Develop methods for objectively prioritizing restoration activities.
- Develop strategies for implementing these habitat restoration priorities.

1.4.1 Restoration Paradigm

In the strictest sense, ecosystem restoration is defined as the establishment of environmental conditions that attempt to duplicate or mimic a pre-disturbance state (National Research Council [NRC], 1992 and 2002). A number of factors in the MRG preclude complete restoration of the Rio Grande. From a biological perspective, the extinction of several aquatic species prevents complete restoration to historic, pre-disturbance conditions. Furthermore, technical, economic, and legal considerations associated with development in the historic floodplain, as well as existing interstate water agreements, limit restoration options from a practical perspective. For this document, the term restoration is used more casually to refer to passive or active processes that promote environmental conditions suitable for the silvery minnow and/or flycatcher (see Section 4.2 for additional discussion of passive and active restoration processes). This functional approach may include practices more accurately categorized as habitat restoration, creation, rehabilitation, replacement, mitigation, enhancement, and naturalization (NRC, 2002). Finally, although this plan focuses on the habitat requirements of the listed species, the Subcommittee recognizes the importance of systemic ecological processes that are needed to maintain ecosystem structure and function. Thus, to the extent practical active or passive restoration techniques that encourage ecosystem functions will be implemented in recognition of the integrated nature of river and riparian systems.

1.4.2 Habitat Restoration Focus Area

The Subcommittee recognizes that the aquatic and riparian habitats of the Rio Grande and Rio Chama are ultimately affected by watershed conditions in the Upper and Middle Rio Grande basins. However, for a number of practical reasons, this plan will focus on activities within a more restricted geographical area. Specifically, the Subcommittee's planning process primarily focuses on the riverine and riparian zones along reaches of the Rio Chama below Abiquiu Dam and the main stem of the Rio Grande from Velarde to Elephant Butte Reservoir. In addition, most, if not all, restoration activities the Subcommittee expects to consider and recommend for funding by the Program will occur between the existing levees in the Program area. One or more planning documents to be developed by the Subcommittee will define reach-specific restoration priorities and activities. All restoration activities supported by the Subcommittee on behalf of the Program will occur within areas consistent with the critical habitat designation of the silvery minnow (FWS, 2003a) and the Final Recovery Plan developed for the flycatcher (FWS, 2002). As indicated earlier, pueblo lands within this area are excluded unless specifically requested or allowed by these entities.

1.4.3 Intent and Relation to Other Plans

Restoration plan and activities have been proposed and implemented by a number of entities. This plan is not intended to supersede existing plans, nor should it prevent the development of future plans or restoration activities by entities outside the Program. Ideally, the Subcommittee hopes that the information contained in this document will facilitate coordinated efforts in the MRG to the benefit of the ecosystem and water users. Because of the integrated nature of watersheds, rivers and riparian areas, a basin-wide and comprehensive planning process is most likely to yield lasting benefits.

1.5 Document Organization and Structure

This plan is intended to provide a framework for developing restoration actions beneficial to the silvery minnow and flycatcher. Overall, the restoration plan is expected to be refined as more information becomes available; thus, it can be considered a living document. This document is the first part of a more comprehensive planning process and provides basic information to help chart the long-term strategy and priorities for habitat restoration in the MRG. Ultimately, it will be augmented by a companion document (or documents) containing detailed information on a reach-by-reach basis to support the development of site-specific plans that can be implemented.

Internally, this document is structured to provide background information on the MRG basin (Section 2), and the ecology and biology of the listed species and their habitat requirements (Section 3). Restoration practices are discussed in Section 4, while Section 5 proposes restoration priorities and a mechanism for evaluating restoration projects.

2.0 PROFILE OF THE MIDDLE RIO GRANDE BASIN

The Rio Grande watershed (355,500 square miles) is the fifth largest in North America, extending 2,000 miles from its headwaters in the San Juan Mountains of Colorado to the Gulf of Mexico near Brownsville, Texas. The following subsections provide descriptions of the climate, geology, hydrology, geomorphology, vegetation, and some aspects of water use (depletions) in the MRG basin. This information is provided as the foundation for understanding the physical and biological attributes of the basin, which may be important for broad scale restoration planning.

2.1 Overview of the Rio Grande and Rio Chama

For the first 70 miles below the Colorado state line, the Rio Grande winds through a deep basalt-lined gorge that is markedly rocky and with limited riparian vegetation. In this area, the Rio Grande flows are augmented by groundwater discharge from beneath the lava-capped plateau and tributaries draining the Sangre de Cristo Mountains. Red River, Rio Embudo, and Rio Pueblo Taos are major east-side tributaries to the Rio Grande. About 200 acres of irrigable land are served by direct diversion from the Rio Grande in the vicinity of Pilar and Rinconada. As the Rio Grande leaves the gorge, the valley begins to widen and alluvial deposits form the banks of the river, with the first appearance of significant areas of riparian vegetation.

Just above Velarde, the Rio Grande flows through the alluvium of the Espanola Valley, and water is diverted to over 5,000 acres of land. The Rio Chama, Santa Clara Creek, Santa Cruz River, and Pojoaque Creek discharge to the Rio Grande in this reach. The Espanola valley contains riparian vegetation and irrigated agriculture. At the south end of the Espanola Valley, the Rio Grande flows past Otowi gage and into the White Rock canyon section containing only minor riparian vegetation until it reaches Cochiti Reservoir.

The Rio Chama enters the Rio Grande north of Espanola. The 40-mile stretch of the Rio Chama between El Vado and Abiquiu Reservoirs is characterized as a relatively narrow and rocky canyon section. Below Abiquiu Dam, the valley widens and there are numerous agricultural diversions to meet local acequia needs. A major Rio Chama tributary, the Rio Ojo Caliente, discharges into the Rio Chama about 6 miles above its confluence with the Rio Grande. Roughly 500 acres of land are irrigated from the Rio Ojo Caliente.

Below Cochiti Dam, the Rio Grande flows into a relatively broad basin and the valley contains an extensive interconnected network of canals and drains. About 60,000 acres of land are irrigated between Cochiti and Elephant Butte Reservoir.

2.2 Regional Climate

Mild, arid to semiarid, continental climatic conditions characterize the MRG basin in New Mexico. Average annual precipitation ranges from less than 10 inches at the valley stations to more than 20 inches at the highest elevations of the basin (Table 2-1). Summer rains are characterized by brief, high-intensity storms coming mainly in July and August. Winter is the driest season with much of the precipitation falling as snow in the mountains and rain or snow in the valleys. Like most arid regions, the basin experiences large variations in seasonal and annual precipitation. Annual precipitation may vary by a factor of 3 or more from year to year (Table 2-1).

Table 2-1: Summary of climatic statistics for selected MRG basin weather stations *

Station	Elevation	Mean Annual Precipitation	Annual Precipitation		Mean Annual Temperature	Period of Record
			Maximum	Minimum		
Taos	6,990	12.42	22.73	6.39	46.88	1914 - 2001
El Vado Dam	6,800	14.28	24.09	6.06	44.65	1923 - 2001
Cuba	6,910	13.04	25.81	6.62	46.52	1941 - 2001
Abiquiu Dam	6,400	9.79	16.58	4.98	51.15	1957 - 2001
Santa Fe	7,000	13.98	21.75	5.03	49.11	1890 - 1972
Cochiti Dam	5,560	12.77	19.86	6.82	54.44	1975 - 2001
Bernalillo	5,040	8.97	16.73	4.39	54.57	1924 - 1982
Albuquerque Airport	5,310	8.56	15.88	3.29	56.58	1914 - 2001
Bernardo	4,730	8.10	13.92	3.39	55.99	1933 - 2001
Socorro	4,620	9.13	17.65	3.03	57.70	1914 - 2001
Bosque Del Apache	4,520	8.81	14.73	2.72	57.87	1914 - 2001
Elephant Butte Dam	4,580	9.19	16.94	3.77	60.97	1917 - 2001

* Data from Western Regional Climate Center: http://www.wrcc.dri.edu/CLIMATEDATA.html.

Mean annual temperature varies from about 61°F in the southern Program area to near 40°F in the highest elevations of the basin. Daytime temperatures often exceed 100°F during the summer below an elevation of about 5,000 feet. The warmest conditions generally occur in June before the thunderstorm season starts. Minimum temperatures below freezing are common throughout the basin in the winter. The freeze-free period ranges from about 200 days at the lower elevations to about 80 days in the northern mountains.

Droughts and periods of above average precipitation occur frequently but with varying degrees of intensity. Severe and prolonged droughts occur on average about once a century (S.S. Papadopulos & Associates, Inc. [SSP&A], 2001). Prolonged wet periods occur about twice each century on average; however, they are generally of shorter duration and lower magnitude than the most severe droughts. Table 2-2 summarizes the estimated lengths and magnitudes of drought and wet periods over the last 500 years. Notably, the period from 1978 to 1992 is considered to represent the wettest conditions in the last 500 years (SSP&A, 2001).

Table 2-2: Summary of extreme regional drought and wet periods in the MRG basin
(SSP&A, 2001)

	Duration (years)	% Average Annual Precipitation
Drought Periods		
1573 – 1593	21	82
1663 – 1677	15	93
1727 – 1743	17	89
1748 – 1759	12	93
1773 – 1783	11	89
1899 – 1904	6	86
1950 – 1964	15	90
Wet Periods		
1507 – 1515	9	104
1608 – 1615	8	104
1633 – 1653	21	108
1720 – 1725	6	104
1764 – 1772	9	107
1790 – 1800	11	104
1830 – 1841	12	108
1852 – 1870	19	112
1882 – 1889	8	107
1905 – 1933	29	109
1978 – 1992	15	123

2.3 Geology

The geology of the MRG basin exerts an important influence on the geomorphology, water quality, and flows in the Rio Grande. The Rio Grande follows a series of deep, interconnected structural depressions (basins) that are referred to collectively as the Rio Grande Rift (Kelly, 1952). The Rio Grande Rift extends more than 600 miles from central Colorado to southern New Mexico and represents one of the major topographic and hydrologic features of the southwestern U.S. (Hawley et al., 1995). The Rio Grande Rift generally coincides with the boundary between two major physiographic provinces: the Colorado Plateau to the northwest, and the Southern Rocky Mountains and Great Plains to the east. The Rio Grande Rift is believed to have developed as a result of extensional (pull apart) forces acting on the earth's crust, which resulted in downfaulting of the rift graben relative to the uplifted mountainous areas along the margins of the rift. Vertical offsets along basin-bounding faults can exceed 5 miles in some locations (Hawley et al., 1995). These faults generally consist of a set of closely spaced parallel faults that stairstep downward into the basin.

The Rio Grande Rift in the MRG includes a number of more or less separate basins that have been designated (from north to south) the San Luis, Espanola, Santo Domingo, Albuquerque, Socorro, La Jencia, and San Marcial basins. Canyons or narrows of varying length separate these basins. The major canyon sections include the Rio Grande Gorge (Taos area) and White Rock Canyon near Los Alamos, while less prominent narrows or constrictions occur at San Felipe, Isleta, San Acacia, and Socorro. Extensive drilling investigations in the Albuquerque area have allowed geologists to further subdivide the rift into the Santo Domingo, Calabacillas, and Belen sub-basins, and intervening structural highs (ridges) (Bartolino and Cole, 2002).

From the Colorado state line to the mouth of Elephant Butte Reservoir, the main channel of the Rio Grande passes through a wide variety of igneous and sedimentary rocks that range from Precambrian to Holocene in age. Detailed geologic maps are available for much of the region (New Mexico Bureau of Geology and Mineral Resources [NMBGMR], 2003). Occurrence and characteristics of some of the more important rock types in the major watersheds are summarized below.

2.3.1 Santa Fe Group

The broader valley sections of the Rio Grande flow through areas underlain by the Santa Fe Group. The downfaulted basins of the Rio Grande Rift are filled with thick deposits of unconsolidated to poorly consolidated sands and gravels, silts, and clays of the Santa Fe Group (Tertiary). Deposition of the Santa Fe Group occurred over the past 30 million years simultaneously with rift subsidence and ceased about 1 million years ago when the Rio Grande began to cut its present valley (Hawley and Haase, 1992). Abundant fine-grained lake sediments in the lower Santa Fe group indicate closed-basin conditions during this early period, but sometime between 3 and 2 million years ago, the Rio Grande became a through-flowing stream, as it remains today (Bartolino and Cole, 2002). Santa Fe Group sediments range up to about 14,000 feet thick in the central part of the basin (Thorn et al., 1993). These basin-fill sediments are important aquifers that yield large quantities of potable groundwater. The Santa Fe Group has been divided into upper, middle, and lower units. The upper Santa Fe Group constitutes the most productive aquifer, contains the highest quality groundwater, and is the most important Santa Fe Group unit with respect to its influence on the river.

Santa Fe Group sediments include alluvium transported and deposited by the ancestral Rio Grande as it migrated back and forth across its floodplain, as well as poorly-sorted alluvial fan sediments eroded from nearby mountains. Lesser quantities of eolian (windblown) and lacustrine (lake) sediments are also encountered in the Santa Fe Group, along with occasional lava flows and tephra beds derived from volcanic eruptions (Bartolino and Cole, 2002). As would be expected, the composition of these sediments strongly reflects the bedrock lithologies from which they were eroded. For example, upper Santa Fe Group sediments in the Albuquerque basin contain large amounts of quartz and feldspar, reflecting the predominant mineralogy of nearby plutonic igneous rocks.

In many areas, younger alluvial sediments (Pleistocene and Holocene) overlie the Santa Fe Group, but because these younger sediments are generally in hydraulic connection with the Santa Fe Group, they are often included as part of the Santa Fe Group aquifer system. Detailed information regarding the geology and hydrogeology of the Santa Fe Group can be found in Thorn et al. (1993), Hawley et al. (1995), and numerous publications by the U.S. Geological Survey (USGS, 2003). The MRG basin as defined by the USGS includes only the reach from Cochiti Dam to San Acacia, whereas the Program area extends from the Colorado-New Mexico state line to Elephant Butte Reservoir.

Major tributaries to the Rio Grande within the study area include the Red River, Rio Chama, Jemez River, Rio Puerco, and Rio Salado. The geology of these watersheds has an important influence on the water quality and sediment load of the Rio Grande.

2.3.2 Red River Watershed

The Red River drains about 190 square miles east of Questa, NM. This watershed includes steep mountainous terrain composed of Precambrian igneous and metamorphic rocks, as well as mineralized Tertiary intrusive and extrusive igneous rocks. Because of sulfide mineralization, the region has a long mining history and numerous natural "alteration scars" are present along the upper part of the watershed. The alteration scars are steep, barren areas that are eroding rapidly as a result of oxidation and weathering of sulfide minerals (e.g. pyrite). Runoff from these scars carries orange iron oxides and clay minerals, giving the Red River its name. As a result of weathering of sulfide minerals exacerbated by both natural processes and mining activities, metals associated with acid rock drainage have been mobilized and transported to the Red River, including aluminum, arsenic, cadmium, chromium, cobalt, lead, iron, manganese, and zinc. Recent studies suggest that aluminum is the primary constituent that limits water quality in the Red River (New Mexico Environment Department [NMED], 2001)

2.3.3 Rio Chama Watershed

The Rio Chama drains about 3,044 square miles above its confluence with the Rio Grande near Espanola. Significant tributaries include Willow Creek and the Rio Brazos, Rio Vallecitos, and Rio Tierra Amarilla. The eastern portion of the Rio Chama watershed includes large areas of mountainous terrain (Brazos and Tusas Mountains) composed predominantly of Precambrian granitic and metamorphic rocks (e.g. quartzite), as well as Tertiary volcanic rocks of the Jemez Mountains. The central and western portions of the watershed are a low-relief basin (Chama and San Juan basins) consisting predominantly of more-easily eroded, fine-grained sedimentary rocks, including sandstones and shales of the Chinle Formation (Triassic), Morrison Formation (Jurassic), Mancos Shale (Cretaceous) and Mesa Verde Group (Tertiary) (Bingler, 1968). Landslides accelerate the erosion of these weak sedimentary rocks on steep slopes, and the resulting sediment is transported to the Rio Chama.

2.3.4 Santa Fe River Watershed

The Santa Fe River drains some 231 square miles above its confluence with the Rio Grande near Cochiti. It is one of the smaller tributaries in the study area and is dry much of the year. The headwaters of the Santa Fe River lie in the crystalline rocks of the Sangre de Cristo Range. On its way to the Rio Grande, the Santa Fe River also cuts through basin-fill sediments, as well as volcanic rocks associated with the Jemez and Cerros del Rio volcanic fields. Flows in the lower part of the Santa Fe River are mainly from the City of Santa Fe wastewater treatment facility, which are impounded behind Cochiti Dam. Flows in the Santa Fe River below Cochiti Dam represent seepage from Cochiti Reservoir.

2.3.5 Jemez River Watershed

The Jemez River drains 1,038 square miles above its confluence with the Rio Grande near Bernalillo. The Jemez River drains the central and southern portion of the Jemez Mountains, as well as the southern part of the Sierra Nacimiento. Important tributaries to the Jemez River include the Rio Guadalupe, which joins the Jemez River near Jemez Springs, and the Rio Salado, which joins the Jemez River near San Ysidro, (not to be confused with the larger Rio Salado near Socorro). The Jemez Mountains are composed predominantly of volcanic tuffs and basalts of Pleistocene age. Because these volcanic rocks are relatively hard and resistant to erosion, the Jemez River above San Ysidro contains little suspended sediment. In contrast, the channel of the Rio Salado is choked with reddish alluvium and loess derived from erosion of soft Paleozoic and Mesozoic sedimentary rocks, particularly shales and mudstones of the Yeso and Chinle Formations. Similar to the much larger Rio Puerco drainage located to the southwest, the upper portion of the Rio Salado is presently downcutting into these soft channel sediments, resulting in a vertical-walled channel that sloughs great quantities of fine sediment during major storm events.

Numerous hot springs, warm springs, and cold springs discharge to the Rio Jemez and its tributaries. While flow rates are generally small, many of these springs discharge water that contains high concentrations of dissolved salts (Summers, 1976). As a result of the elevated salinity, white evaporite salts precipitate on the channel sediments near San Ysidro during low flow periods, and these salts are redissolved during high flows and transported to the Rio Grande. The gypsiferous Todilto Formation is likely another contributor to the salinity that is prevalent in this area. Geothermal springs that discharge to the Rio Jemez often contain elevated concentrations of arsenic. As a result, the arsenic concentration in the Rio Grande rises significantly below its confluence with the Rio Jemez.

2.3.6 Rio Puerco Watershed

The Rio Puerco drains a large area of Sandoval, Cibola, McKinley, Bernalillo, and Valencia Counties (7,350 square miles), and merges with the Rio Grande near Bernardo. The Rio San Jose is a major tributary of the Rio Puerco, and drains the area from Grants to Laguna. The Rio Puerco is an ephemeral stream for most of its length, flowing only in response to significant storm events. The Rio Puerco is famous for its large sediment loads, and it has been estimated that this tributary contributed more than 50 percent of the sediment load to the Rio Grande in central New Mexico, but only about 16 percent of the water (Fox et al., 1995). Along much of its length, rapid downcutting since the 1880s has transformed the Rio Puerco from its former wide, shallow channel to the present narrow channel with high banks. The channel has also migrated laterally, and it is reported that more than 90 percent of the lower channel has shifted position since 1954 (Young, 1982). Improper grazing is often attributed with triggering channel incision in the Rio Puerco in the late1800s: however, there is evidence that the valley of the Rio Puerco has undergone many alternating cycles of downcutting and infilling over the past several million years (Fox et al., 1995). Today, the systemic downcutting in the Rio Puerco watershed has subsided and localized backfilling is evident (Elliott et al., 1999).

The geomorphology of the Rio Puerco is intimately linked to the surficial geology. While volcanic and plutonic igneous rocks are present in the watershed, the sediment load carried by the Rio Puerco results primarily from erosion of extensive outcrops of soft, fine-grained Paleozoic and Mesozoic sedimentary rocks. Shales and mudstones of the Yeso, Chinle, Morrison, and Mancos Formations contribute great quantities of clay and silt to the Rio Puerco, and ultimately the Rio Grande, during storm events.

2.3.7 Rio Salado Watershed

The Rio Salado drains some 1,380 square miles above its confluence with the Rio Grande near Bernardo. This watershed includes a wide range of rock types, including Tertiary volcanic rocks of the Datil Mountains, Precambrian metamorphic rocks of the Ladron Mountains, and sedimentary rocks (limestone, mudstone, gypsum) of the Sierra Lucero Uplift. Perhaps the most widespread are the Chinle, Mancos, and Mesaverde shales and mudstones. These units are easily erodible, and surface runoff flowing over unvegetated areas may transport large quantities of suspended sediment. The hard, competent volcanic and metamorphic rocks are sources for coarse sediment contributions to the Rio Grande.

2.4 Hydrology

Components of the silvery minnow and flycatcher habitat are influenced by hydrologic conditions. The silvery minnow is an aquatic organism, while flycatcher habitat is associated with riparian communities. Beyond these simple concepts, the relationship of hydrology to habitat for these species is complex and poorly characterized. Because water in the MRG basin is considered to be fully appropriated and a large portion of the river flow must be delivered to Texas under the Compact, changes in the amount of water needed to maintain restored areas or to achieve mandated target flows have important societal, economic, and ecological consequences. Ultimately, if additional water is needed for endangered species, an existing water use must be suspended. The Subcommittee believes that the Program should investigate the possibility of flexibility in management of flows required for Compact deliveries in terms of potential benefits to the habitat restoration features developed. The timing, duration, and quantity of flows associated with the Program features are important considerations to address water use and optimize success. The intent of this section is to provide an overview of the hydrology of the MRG with emphasis on contemporary conditions including a summary of the institutional constraints associated with the water management facilities.

2.4.1 Historical Hydrology

Stream gaging began in New Mexico in 1888 with the establishment of a USGS training camp on the Rio Grande near Embudo. A gaging station was built near the camp and the collection of continuous streamflow records commenced on January 1, 1889 (Borland, 1970). Prior to that time, our understanding of the hydrology of the MRG is based on anecdotal accounts and inferences from the archeological record and historical documents. Review of the historical hydrology is meant to provide insights into the conditions in which the silvery minnow and flycatchers evolved, rather than to define the goals for restoration based on pre-European conditions. In this light, interpretations of the historical record must consider that humans have modified components of the MRG hydrology for at least the last 400 years (Scurlock, 1998). Secondarily, the historical accounts must be considered in context. For instance, General Diego de Vargas in his 1692 trip to the Pecos area described New Mexico's climate "as so very cold with abundant snow and rain and such heavy frost and freezes." (Scurlock, 1998).

The historical record suggests that the native flows were similar to those seen over the past 110 years. Historically, flows in the Rio Grande were generally perennial, except during periods of drought (Scurlock, 1998). Peak flows most often occurred in late spring in response to snowmelt runoff, while episodic floods occurred in association with late-summer monsoons. Tree-ring and historical evidence suggests that severe and prolonged droughts occur two to three times each century (see Section 2.2). Numerous instances of channel drying are contained in historical reports with the first notation in 1752 (Scurlock, 1998). The frequency of reports of channel drying increased in the late-1800s as agricultural irrigation in the upper and middle basins became more prevalent.

References to floods are common in the historical record. Scurlock (1998) estimated that between 1849 and 1942 about 50 moderate to large floods (greater than 10,000 cubic feet per second [cfs]) occurred in the MRG. Wozniak (1998) reported that the flood of 1874 destroyed almost every building between Alameda and Barelas. The communities of Tome, Valencia, and Belen were under water during the spring flood of 1884. Tome was later washed away in a 1905 flood (Wozniak, 1998). Property damage and loss of agricultural capacity associated with floods prompted the establishment of flood control measures in the early 1900s. The 1941 flood was the last catastrophic flood in the MRG to date. This event flooded parts of Albuquerque and inundated the town of San Marcial and provided the impetus for continued federal intervention, including initiation of the Middle Rio Grande Project (Section 2.4.3).

2.4.2 Early Development in the Rio Grande Valley

Irrigated agriculture has a long history along the Rio Grande, possibly preceding the Spanish Colonization in the mid 1500s. However, Wozniak (1998) indicates that irrigation played a role in the Puebloan subsistence prior to the Spanish reconquest in 1692. Thereafter, irrigation increased in importance for the inhabitants of the Rio Grande Valley. The early irrigation systems, with head works constructed of logs, brush, and stones, were rudimentary, inefficient, and required frequent replacement. Labor requirements to build and maintain the irrigation infrastructure were substantial, and expansion of the irrigation system was in response to population growth (Wozniak, 1998). Prior to the late 1800s, the development of irrigated agriculture had significant implications from a societal perspective, but only minor or localized impacts on the hydrology and geomorphology of the Rio Grande. Irrigation continued to expand into the American territorial period from 1854 to 1910, with punctuated growth and impacts starting in the late 1880s.

The first systemic incursion on the Rio Grande floodplain was associated with the construction of the New Mexico and Southern Pacific Railroad, which began grading at Albuquerque in the winter of 1879-1880. The tracks reached San Marcial on September 10, 1880, and the first regular train was run from Albuquerque to San Marcial on October 1, 1880. Bridges at San Marcial and just south of Albuquerque locally prevented river channel movement, while the railroad embankments isolated portions of the historical floodplain from all but the largest floods. Areas where the railroad significantly encroached on the floodplain included San Marcial, Socorro, San Acacia, La Joya, Tome, Isleta, Corrales, Algodones, and San Felipe. Happ (1948) recognized preferential aggradation of the floodplain on the river side of the railroad embankments.

The first major channelization of the Rio Grande may have been associated with the railroad. Near San Acacia, the Rio Grande channel was shortened about 1.5 to 2 miles to avoid the construction of an additional bridge. Happ (1948) estimated the channel gradient was locally steepened 7 to 10 feet, followed by headcutting, and ultimately adjustment of grade associated with aggradation. A detailed historical evaluation of the channel modifications associated with the railroad construction is lacking, but may provide a more complete picture of the channel conditions prior to the 1917-1918 survey. Indirectly, the railroads provided a means for transporting goods, thus promoting expansion of agricultural enterprises beyond the demand associated with local consumption.

The most pronounced expansion of irrigated agriculture in the late 1800s was in the Upper Rio Grande basin (San Luis valley) where the nominal area under irrigation in 1870 escalated to more 300,000 acres over the next two decades (National Resources Committee, 1938). By the 1890s, irrigation diversions in the San Luis valley in Colorado reduced native flows in the river by 40 to 60 percent (National Resources Committee, 1938). Wozniak (1998) described the conditions in the MRG during this time:

> By the early 1890s, serious problems had emerged in the irrigation agriculture of the Rio Grande Valley. Drought, which had struck sporadically in the 1880s, became acute in the 1890s (Baker 1898:17-18; Wortman 1971:17); by 1898 the Rio Grande below Albuquerque literally dried up for 4 months of the year. Stream flow had been seriously depleted by rapid development of irrigation agriculture in the San Luis Valley of Colorado; the effects on downstream users were dramatic and ultimately led to federal intervention (Follett 1896; Harper et al. 1943; Harroun 1898; Yeo 1910, 1929).
>
> Ironically, at the same time that the Rio Grande was being seasonally depleted, lands in the middle Rio Grande Valley from Cochiti to San Marcial, especially between Bernalillo and La Joya, were becoming waterlogged and thus not amenable to cultivation (Conkling and Debler 1919:77; Harper et al. 1943; Harroun 1898:2; Natural Resources Committee 1938:70). Sedimentation in the Rio Grande resulting from decreased flows had caused the bed of the main channel to aggrade; as a result the water table in the many parts of the valley had begun to rise. Waterlogged lands had always been a problem near the Rio Grande owing to poor drainage and wasteful irrigation practices; under traditional agricultural methods, excess water in the acequias was simply dumped onto the low-lying lands at the end of the acequia. Only a small percentage of the ditches had facilities for returning the excess flow to the Rio Grande or delivering the water to downstream ditches. Each ditch system, of which there were dozens, was independent; no plan or organization to integrate the multitude of irrigation systems in the middle Rio Grande Valley existed or was deemed necessary.
>
> As early as the 1820s, local farmers had noted the formation of marshes in the Rio Grande Valley that were the result of the dumping of excess flows from ditches (CPLC, Case 8; CPLC, Case 51). In the 1880s and 1890s, the aggrading of the Rio Grande and rising water table had exacerbated the situation.

Agricultural conditions in the MRG continued to decline into the early 20th century. Floods, aggraded channel conditions, poorly drained soils, salinization, and water shortages associated with drought and the upper basin irrigation demands limited agricultural development in the MRG during the late 1800s and early 1900s (Natural Resources Committee, 1938; Wozniak, 1998). Sediment loads to the Rio Grande from MRG tributaries (e.g., Chama, Jemez, Galisteo, Rio Puerco, and Rio Salado) were elevated in the late 1800s by arroyo incision and changes in land use in the basin (Happ, 1948). Major increases in sediment load occurred downstream of the Rio Puerco confluence as a result of channel incision associated with drought and watershed degradation (Rittenhouse, 1944; Happ, 1948; Elliott, 1979; Scurlock, 1998).

From 1880 to 1925, the amount of cultivated land was estimated to range from about 32,000 to 50,000 acres (Wozniak, 1998). Hedke (1925) is often cited as indicating that 125,000 acres of lands were under irrigation in 1880; however, Wozniak (1998) explains that this value was inflated to exaggerate the tragedy of loss in the MRG Valley.

> A careful examination of the data upon which Hedke relied shows that the figure of 125,000 irrigated acres was largely conjectural, a projection of the acreage that could have been served by the ditches that probably existed in 1880. It was not an actual estimate of cultivated lands.
>
> Even Hedke (1925) indicated that only about 44,000 acres were actually irrigated in 1880. By 1896, as a result of development in the San Luis Valley and a decade-long drought, irrigated acreage had fallen to approximately 32,000 acres in the middle Rio Grande Valley (Follett 1896). Irrigated acreage rebounded to approximately 45,000 acres by 1910 (Yeo 1910), to 48,000 acres in 1918, and to almost 49,000 acres in 1923 (Gault, 1923), then fell to around 40,000 acres in 1925 (Hedke 1925). This last decline had as much to do with the collapse of agricultural prices after World War I as it did with the deterioration of the physical resources in the middle Rio Grande Valley, although the latter certainly occurred as well.

It is clear from the preceding discussions that the hydrologic and geomorphic conditions in the MRG were substantially modified by the early 1900s. Thus, comparison to the conditions that existed in the early 1900s as natural references must be made with caution.

2.4.3 Reclamation of the Middle Rio Grande

Water storage, drainage, and flood control projects initiated in the 1930s allowed increased agricultural development, and currently about 60,000 acres are cultivated in the MRG. Renovation of the irrigation system in the MRG began in the 1930s, following the establishment of the MRGCD in 1925 (Wozniak, 1998). El Vado dam on the Rio Chama was completed in 1935. By 1936, the renovation included construction of four new diversions (Cochiti, Angostura, Isleta, and San Acacia) that replaced numerous old diversions: a siphon at Corrales, and two diversion headings at Atrisco and San Juan. In addition, 767 miles of canal were constructed or rehabilitated along with 342 miles of drains and 180 miles of riverside levees (Wozniak, 1998). Although initially a success, the floods of 1941 damaged the infrastructure of the MRGCD and, with the Great Depression, led to its temporary inability to cope with the MRG's water and management challenges. In 1942, the Corps and Reclamation began investigating the problems in the District, and ultimately intervened through the Middle Rio Grande Project under the Flood Control Acts of 1948 and 1950 (Wozniak, 1998).

Under the Middle Rio Grande Project, Reclamation was primarily responsible for stabilizing and improving the Rio Grande channel to increase the efficiency of water conveyance, financial restructuring of the MRGCD, rehabilitation of the MRGCD irrigation and drainage systems, and water conservation. Under subsequent authorizations of the Middle Rio Grande Project, the Corps was assigned primary responsibility for flood control, including authorizations to build levees on the Rio Grande and flood control reservoirs on the Rio Chama, Jemez River, Galisteo, and the main stem of the Rio Grande at

Cochiti. Reclamation's rehabilitation program consisted of repairs to El Vado Dam, Cochiti, Angostura, Isleta, and San Acacia diversions; improvement of the canals, laterals, and drains; channelization of 127 miles of river above Elephant Butte Reservoir; channel rectification in the Española Valley and Hot Springs (i.e., Truth or Consequences) reach; and construction of the LFCC (Wozniak, 1998). Maintenance and channel rectification projects continued throughout the 20th century and are still ongoing. The activities undertaken by Reclamation and the NMISC improved agricultural conditions and allowed New Mexico to meet the Compact delivery obligations.

2.4.4 Contemporary Surface Water Hydrology

The MRG basin is centered in a semiarid region where potential evaporation far exceeds precipitation. On average, about 1.1 million acre-feet (AF) of water passes Otowi gage each year. Water is supplied to the Rio Grande about equally from the upper basin in Colorado, and from the Sangre de Cristo Mountains and Rio Chama watersheds in New Mexico. Consistent with the climate of this region, native flows are subject to significant variability. For example, over the last century annual mean streamflow at the Otowi gage ranged from about 495 to 3,580 cfs. There is a trans-basin diversion from the San Juan River basin to the Rio Grande associated with the San Juan Chama Project (SJCP), which supplies about 100,000 AFY. Native flows refer to waters that originate within the Rio Grande basin. SJCP water is not considered native water.

The Rio Grande is a losing river through most reaches (Fig. 2-1). Water in the river is lost to surface evaporation, diversion, evapotranspiration (ET) by riparian vegetation and agricultural crops, and groundwater recharge to riverside drains, the LFCC, and the deeper aquifer. Tributary inflows, irrigation return flows and treated municipal wastewater locally augments flow in the river.

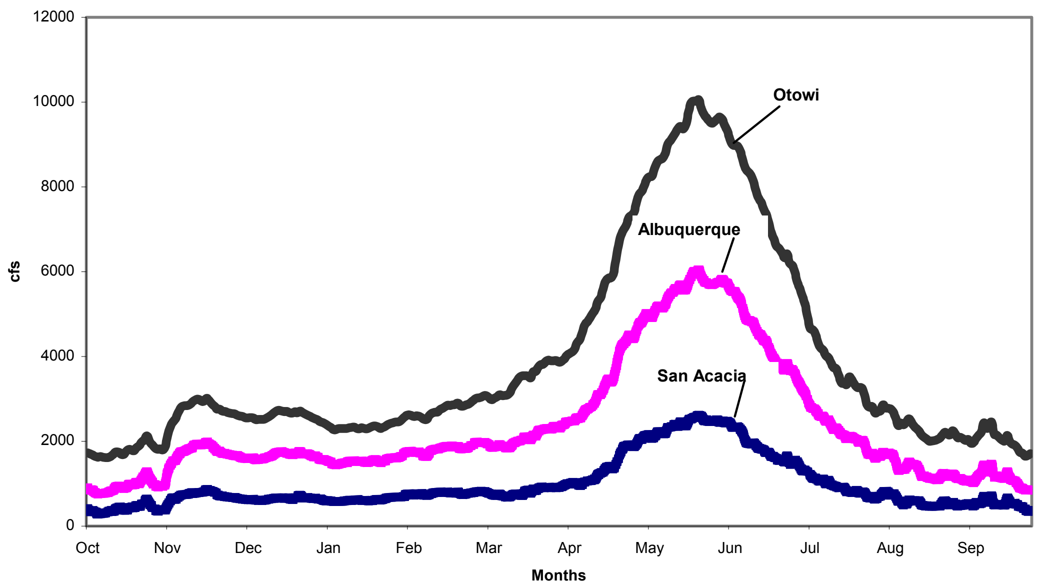

Figure 2-1: Average mean daily flow for the period 1971 to 1999

Comparison of pre-Heron Reservoir (pre-1971) and post Cochiti Reservoir flows (1974-1997) (Figs. 2-2, 2-3, and 2-4) indicates that median daily (i.e., 50 percent exceedance) flows between Otowi and Albuquerque have increased by approximately 200 to 300 cfs (Mussetter Engineering, Inc. [MEI], 2002). The increases in the median daily flow are notably higher in the post-Cochiti Reservoir period for the Bernardo, San Acacia, and San Marcial gages (MEI, 2002). Thus, flow in the river was usually more consistent during the last quarter of the 20th century. The increased flows resulted from a number of factors including the importation of SJCP water, changes in the operation of the Rio Grande Conveyance Channel and LFCC, discharges from the City of Albuquerque wastewater treatment plant (about 69,000 acre feet per year [AFY]), and operation of the flood control dams (MEI, 2002). The relatively wet climate of the 1980s and 1990s also contributed to the observed increase in flow during the post-Cochiti Reservoir period (Section 2.2).

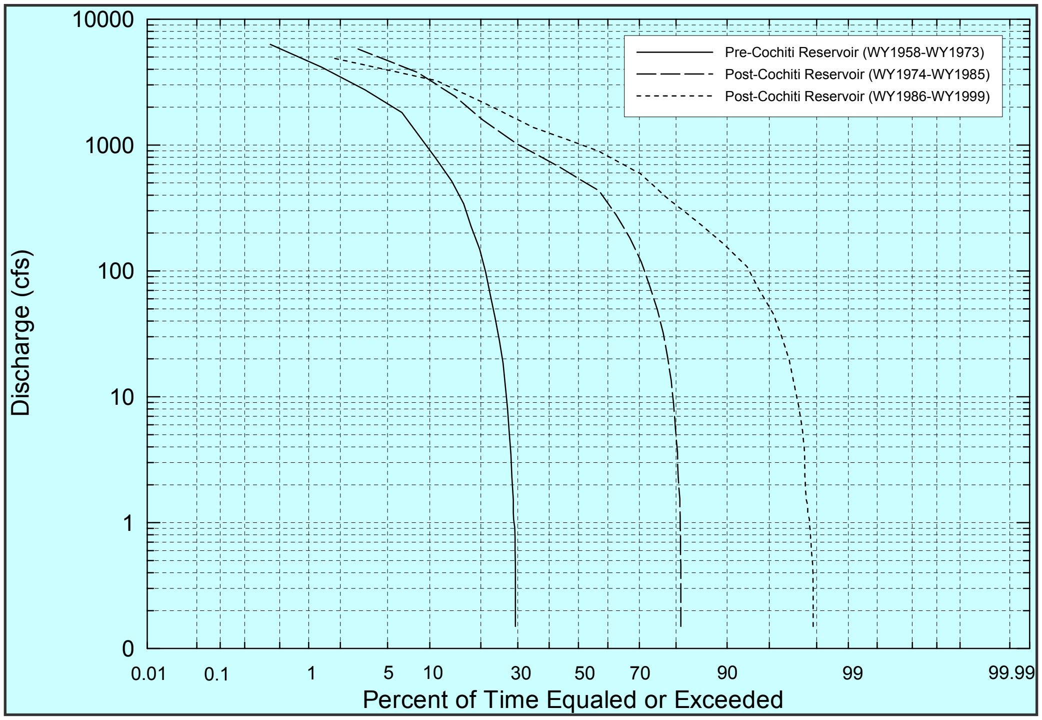

Figure 2-2: Flow-duration curves for the Rio Grande near Bernardo
(MEI, 2002)

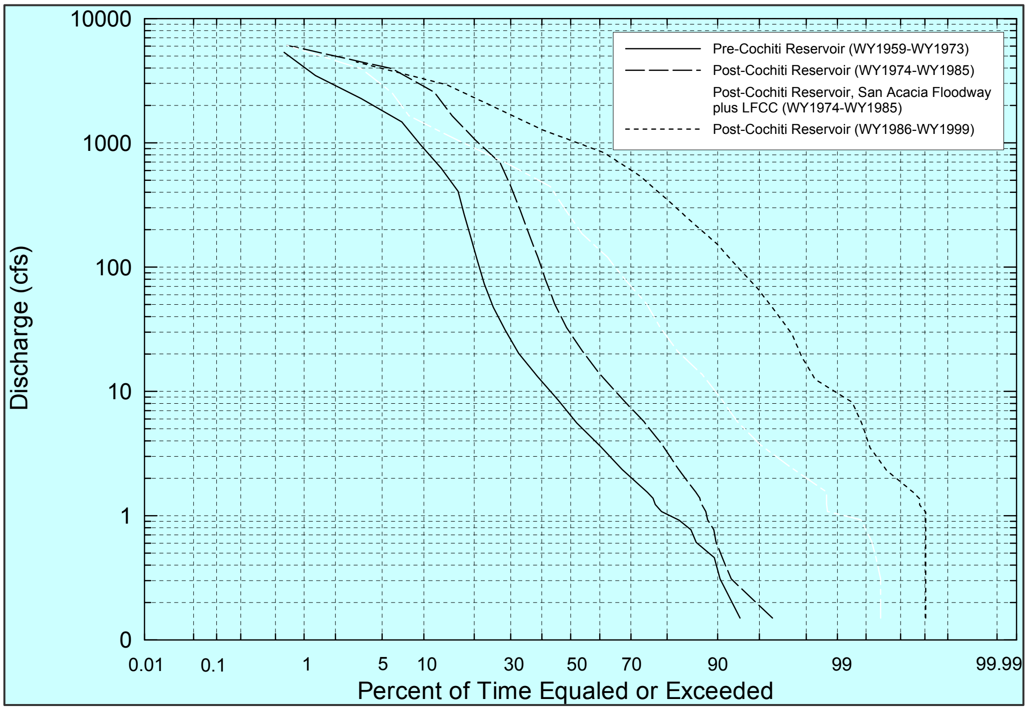

Figure 2-3: Flow-duration curves for the Rio Grande at San Acacia
(MEI, 2002)

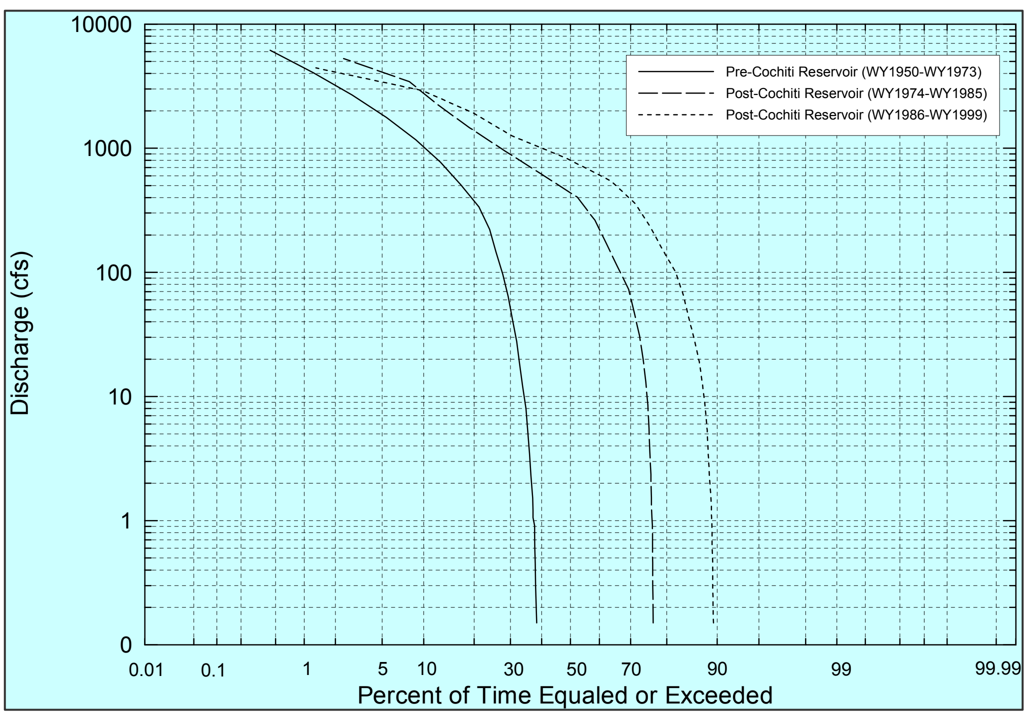

Figure 2-4: Flow-duration curves for the Rio Grande near the San Marcial gage
(MEI, 2002)

Channel drying is rare between Cochiti Dam and Isleta but is more common in the Bernardo, San Acacia, and San Marcial reaches. Analysis by MEI (2002) of USGS gage data indicates that the river was dry about 70 percent of the time at Bernardo between 1958 and 1973, but only about 3 percent of the time in the 1986-1999 period. At San Acacia, the river was dry about 10 percent of the time in the earlier period (1958-1973) and less than 1 percent of the time in the latter period (1986-1999). The longer duration of flow at San Acacia than Bernardo was probably the result of irrigation return flows that enter the river below Bernardo and/or the restriction of the valley at San Acacia. In either case, the flows at San Acacia were minor and probably did not persist for a significant distance downstream of the gaging station. The difference in the duration of flow between these two periods was probably related to a combination of flood control operations at Cochiti Dam and abnormally wet conditions in the 1980s and 1990s. In addition, the operation of the LFCC between 1960 and 1985 affected flows below San Acacia.

The maximum daily mean flows typically occur in the spring in association with snowmelt runoff from the upper basin in Colorado, Sangre de Cristo Mountains, and the Rio Chama watershed (Fig. 2-1). Peak flow events typically occur during April and May, although in any given year the peak event may occur in June, July, and August or more rarely in September and October. Annual peak flow events tend to occur more frequently in the late summer (July and August) in the lower reaches and in the spring (April and May) in the upper reaches of the MRG. Sustained high volume flows are more likely to occur in the spring rather than in the summer months. The last major floods on the Rio Grande occurred in 1941 and 1942, with flows of about 25,000 cfs recorded at the Bernalillo and Albuquerque gages. The largest flow on record in the MRG was 47,000 cfs at the San Marcial gage in September 1929.

Flood control operations at Abiquiu, Cochiti, Galisteo, and Jemez Canyon Reservoirs reduce peak flows below Cochiti Dam in some years. Cochiti Dam releases are restricted to the maximum nondamaging downstream channel capacity, which was typically estimated to be 7,000 cfs at the Albuquerque gage. However, recently releases have been further constrained by conditions of selected levees and the railroad bridge at San Marcial (C. Gorbach, Reclamation). Flood control operations at Cochiti Dam are coordinated to account for flow emanating from Galisteo Creek and Jemez River. Peak streamflow at the Albuquerque gage exceeded 7,000 cfs in 13 of the 29 years (45 percent) between 1942 and 1971 (prior to construction of Cochiti Dam), while peak flows exceeded 7,000 cfs in 53 of 97-years at the Otowi gage (55 percent). Thus, based on this crude analysis, current flood control operations result in attenuation (reduction in size) of large peaks, but have had little effect on the peak hydrograph in about half the years.

The Rio Puerco and Rio Salado are major tributaries that contribute to late summer peak flows in the lower reaches of the MRG. A maximum peak flow of 18,800 cfs was measured on the Rio Puerco and 36,200 cfs on the Rio Salado. These drainages are uncontrolled and large scale flooding can be expected in the futurein the lower reaches.

2.4.5 2003 BO Water Operations Requirements

The March 17, 2003, BO established a set of RPAs addressing water operation elements to be followed during wet, average, and dry years and hydrology requirements for habitat improvement. A dry year is defined as when the Natural Resource Conservation Service (NRCS) April stream flow forecast at Otowi Gage is less than 80 percent of average, and a wet year is one where the April forecast is for 120 percent or higher than average. Average water years are between these extremes, with the average flow defined by the NRCS as being the average streamflow at a point of reference (here, Otowi Gage) for the 30-year period from 1971 through 2000 (FWS, 2003a). The water operation elements are listed below in accordance with the letter designations used in the BO.

Water Operation Elements

A) Spawning Spike - Between April 15 and June 15 of each year, the action agencies, in coordination with parties to the consultation, shall provide a one-time increase in flows (spawning spike) to cue spawning. The need for, timing, magnitude, and duration of this flow spike will be determined in coordination with the Service.

B) Maximum Persisting Habitat Reach - In coordination with the Service, Reclamation and the Corps shall release any supplemental water in a manner that will most benefit the listed species, i.e., produce the maximum persisting habitat reach.

C) Channel Desiccation and Minnow Rescue - Reclamation, in coordination with parties to the consultation, shall conduct routine monitoring of river flow conditions when flows are 300 cfs or less at San Acacia, and shall report information regularly to the Service through the water operations conference calls and meetings.

D) Pumping for Active Flycatcher Territories - Reclamation, in coordination with parties to the consultation, shall ensure that active flycatcher territories supported by pumping from the LFCC are provided with surface water or moist soils in the Rio Grande from June 15 to September 1.

Dry years and/or when storage restrictions from Article VI and/or VII of the Compact are in effect

E) Continuous River Flow from November 16 to June 15 - Action agencies, in coordination with parties to the consultation, shall provide continuous river flow from Cochiti Dam to the southern boundary of silvery minnow critical habitat from November 16 to June 15.

F) 100 cfs at the Central Bridge gage - Action agencies, in coordination with parties to the consultation, shall provide year-round continuous river flow from Cochiti Dam to Isleta Diversion Dam with a minimum flow of 100 cfs at the Central Bridge gage.

G) Managed River Recession - Reclamation shall pump from the LFCC as soon as needed to manage river recession. The pumping capacity must meet or exceed the total capacity of pumps used in the 2002 irrigation season (150 cfs). Pumping shall continue when it will benefit the flycatcher and its habitats.

Average Years

H) Continuous River Flow from November 16 to June 15 - Action agencies, in coordination with parties to the consultation, shall provide continuous river flow from Cochiti Dam to the southern boundary of silvery minnow critical habitat from November 16 to June 15.

I) Ramping Down the Flows - Action agencies, in coordination with parties to the consultation, shall, from June 16 to July 1 of each year, ramp down the flow to achieve a target flow of 50 cfs over San Acacia Diversion Dam through November 15.

J) Year-round Continuous Flow from Cochiti to Isleta - Action agencies, in coordination with parties to the consultation, shall provide year-round continuous river flow from Cochiti Dam to Isleta Diversion Dam with a target flow of 100 cfs over Isleta Diversion Dam.

K) LFCC Pumping - Reclamation shall pump from the LFCC if needed to manage river recession and maintain connectivity. The pumping capacity must meet or exceed the total capacity of pumps used in the 2002 irrigation season (150 cfs). Pumping shall continue when it will benefit the flycatcher and its habitats.

Wet Years

L) Continuous River Flow from November 16 to June 15 - Action agencies, in coordination with parties to the consultation, shall provide continuous river flow from Cochiti Dam to the southern boundary of silvery minnow critical habitat from November 16 to June 15, with a target flow of 100 cfs at the San Marcial Floodway gage.

M) Ramping Down the Flows - Action agencies, in coordination with parties to the consultation, shall, from June 16 to July 1 of each year, ramp down the flow to achieve a target flow of 100 cfs over San Acacia Diversion Dam through November 15.

N) Year-round Continuous Flow from Cochiti to Isleta - Action agencies, in coordination with parties to the consultation, shall provide year-round continuous river flow from Cochiti Dam to Isleta Diversion Dam with a target flow of 150 cfs over Isleta Diversion Dam.

O) LFCC Pumping Reclamation shall pump from the LFCC if needed to manage river recession and maintain river connectivity. The pumping capacity must meet or exceed the total capacity of pumps used in the 2002 irrigation season (150 cfs). Pumping shall continue to maintain river connectivity.

Habitat Improvement Elements

V) Overbank Flooding - Each year that the NRCS April 1 Streamflow Forecast is at or above average at Otowi and flows are legally and physically available, the Corps shall bypass or release floodwater during the spring to provide for overbank flooding. The overbank flooding will be used to create an increased number of backwater habitats for the silvery minnow and flycatcher. The timing, amount, and locations of overbank flooding will be planned each year in conjunction with the Service and may be conducted in coordination with Compact deliveries.

2.4.6 Institutional Constraints

Beyond the limitations imposed by the climate of this region, water operations are controlled by complex institutional and legal constraints associated with authorizations of upstream facilities and diversions from the river. The principal constraint on water use in the MRG is defined by the Rio Grande Compact, negotiated and signed by the states of Colorado, New Mexico, and Texas, enacted as Public Act No. 96 by the 76th Congress and subsequently ratified by each state's legislature and approved by President Roosevelt in 1939. The purpose of the Compact was to equitably apportion the waters of the Rio Grande among the states of Colorado, New Mexico and Texas, based on how that apportionment existed in 1929. The overarching goal of the Compact was to allow each state to develop its water resources at will, subject only to the delivery obligations set forth in the Compact. The Compact is administered by the Rio Grande Compact Commission, which consists of a Commissioner from each of the three signatory states, plus a federal representative appointed by the President who acts as Chairman of the Commission without vote. The portion of the Compact water allocated to the MRG basin is considered to be fully appropriated. Any new use of water, such as the depletions that result from specified endangered species flow requirements, must come from some current existing use. The sustained, long-term use of Compact delivery water to meet specified flow requirements for endangered species would cause additional depletions on the system that could result in the eventual violation of the Compact by New Mexico.

Congress authorized the Middle Rio Grande Project with the Flood Control Acts of 1948 and 1950 with major goals being the reduction of natural depletions, improvement of water delivery to Elephant Butte Reservoir, flood control, and improvement of the drainage and irrigation system. A portion of the project consisted of the construction of the Rio Grande Floodway between Velarde and Caballo Reservoir which included levees, bank stabilization (jetty jack fields), clearing (removal of islands, sandbars and vegetation), and channelization of parts of the river. The LFCC was completed by Reclamation as part of the MRG Project in 1959.

The Corps completed construction of Jemez Canyon Reservoir on the Jemez River in 1954, Abiquiu Reservoir on the Rio Chama in 1963, Galisteo Reservoir on Galisteo Creek in 1970, and Cochiti

Reservoir on the Rio Grande in 1975. These flood control reservoirs were not authorized for conservation storage.

In 1971, Reclamation completed the SJCP with the construction of Heron Reservoir on Willow Creek, a tributary of the Rio Chama above El Vado Reservoir. The SJCP has a firm yield of 96,200 acre feet (AF) of water diverted from three tributaries of the San Juan River in southwest Colorado (Navajo, Little Navajo and Blanco rivers) and stored in Heron Reservoir. On average, about 54,600 AFY of SJCP water has passed Otowi for delivery to contractors in the MRG since the inception of the project.

Heron Reservoir, on the Rio Chama, has a maximum capacity of 401,320 AF and is authorized to store only SJCP water, and all native water is bypassed (about 100 AF per month). Release of stored SJCP water occurs only at the request of contract holders or the New Mexico State Engineer to offset depletions of native Rio Grande water caused by pumping downstream. Carryover storage of SJCP water is not allowed in Heron Reservoir, and all allocated water must be released each year by December 31 or it reverts to the project pool.

El Vado Reservoir on the Rio Chama is operated to maximize the storage of native water during periods allowed under the Compact and to meet the purposes of the 1951 contract to guarantee delivery of irrigation water to the prior and paramount lands of the six MRG pueblos. El Vado Reservoir can store SJCP water. It is currently operated by Reclamation under agreement with the MRGCD. Reservoir releases are made to augment native flows of the Rio Grande in the MRG for irrigation. Typically, the MRGCD uses the direct flow of the Rio Grande during spring run-off and, concurrently, attempts to fill El Vado Reservoir on the Rio Chama. When the native flow of the Rio Grande is insufficient for the MRGCD's diversion needs, releases can be made from El Vado Reservoir. On average, about 15,000 AF of annual storage is used to ensure delivery in accordance with the paramount lands of the six MRG pueblos.

Abiquiu Reservoir, on the Rio Chama downstream from El Vado Reservoir, is authorized to operate for flood control, sediment retention, and storage. Its maximum storage is 1,212,000 AF, with authorizations of 77,000 AF for sediment control and 502,000 AF for flood control. The City of Albuquerque and other entities that contract for SJCP water can store up to 200,000 AF of water in accordance with a contract between the entities and the Corps. When storage of SJCP water is not required, up to 200,000 AF of native Rio Grande water may be stored subject to permitting by the New Mexico State Engineer and consent to a deviation from the Rio Grande Compact Commission. Under current operations, normal releases from El Vado are passed through Abiquiu Reservoir with little or no regulation. Because of channel capacity constraints associated with public safety, reservoir discharges are limited to 1,800 cfs directly below the dam, 3,000 cfs at the Chamita gage, and 10,000 cfs at the Otowi gage.

Cochiti Reservoir provides flood protection from flows on the Rio Grande and Santa Fe River. Its initial authorization included only 105,000 AF for sediment control and 500,000 AF for flood control; storage of native water for a permanent pool was specifically prohibited, although water from outside of the Rio Grande basin can be stored. Subsequently, Congress authorized a 1,200-acre recreational pool, using about 50,000 AF of SJCP water. An annual allocation of 5,000 AF of SJCP water, originally a portion of the City of Albuquerque's annual allocation, was reserved to replace water evaporated from this pool. No part of this project is allocated to irrigation or other uses. Floodwaters are stored only for the duration needed and are released as downstream channel conditions permit; however, if flow at the Otowi gage is less than 1,500 cfs, any floodwater stored in Cochiti Reservoir on July 1 is not released until November 1. Releases from Cochiti are coordinated with the operations at the Galisteo and Jemez Canyon dams to restrict flows at the Albuquerque gage to safe levels.

Galisteo Reservoir on Galisteo Creek, which has its confluence with the Rio Grande about 18 miles downstream of Cochiti Dam, is authorized only for flood control (79,600 AF) and sediment control (10,200 AF). The dam passes all floods up to 5,000 cfs. Normally, this reservoir is dry.

Jemez Canyon Reservoir is on the Jemez River about 3 miles from the Rio Grande about 24 miles downstream from Cochiti Dam. Its authorization includes 40,100 AF for sediment control and 73,000 AF for flood control. The Corps evacuated water from the sediment pool in October 2001. Non-flood native flows typically pass the dam with little or no regulation.

After the 1941 flood, accumulated channel sediment significantly impaired flow in the Rio Grande and adversely affected the delivery of water from New Mexico to Texas, as required by the Compact. To address these issues, Reclamation implemented a program of drainage improvements and channel stabilization, including construction of the LFCC to convey water from the San Acacia diversion dam to the narrows at Elephant Butte Reservoir. The LFCC has also improved drainage, supplemented irrigation water supply, and provided a dependable year-round water supply to the Bosque del Apache National Wildlife Refuge. The LFCC functioned at full design capacity for about 15 years and played an important part in allowing New Mexico to overcome an accumulated debt under the Compact of over 500,000 AF. Diversions into the LFCC were suspended in March 1985 due to the obstruction of the lower portions by sediment; with minor exceptions, the LFCC has carried only drainage and irrigation return flows since then. Since 1985, all flows below San Acacia have essentially been routed down the floodway (river).

2.4.7 Groundwater Hydrology

The MRG basin consists of a number of more or less separate structural basins along its length that have been designated (from north to south) the San Luis, Española, Santo Domingo, Albuquerque, Socorro, La Jencia, and San Marcial basins. The basins are separated from each other by canyons or narrows of varying length, including the Rio Grande Gorge (Taos area), White Rock Canyon near Los Alamos, and narrows or constrictions at San Felipe, Isleta, San Acacia, and Socorro. The Albuquerque basin has been further subdivided into three discrete sub-basins that are separated from one another by structural barriers (e.g., benches and upthrown blocks) or low conductivity pre-basin fill deposits. From north to south these sub-basins include the Santo Domingo, Calabacillas, and Belen sub-basins. Each basin and sub-basin has a somewhat unique structural and depositional history relative to the other basins, and drainage was internal to each sub-basin during the early history of the rift valley. As a result, no major hydrostratigraphic unit extends across all three sub-basins in the deeper parts of the aquifer system (Bartolino and Cole, 2002).

Most significant water-bearing units of the MRG basin are contained within unconsolidated deposits of the Santa Fe Group and Post-Santa Fe Group valley fill deposits. Because these two deposits are hydraulically connected, they are commonly grouped together as the Santa Fe Group aquifer system. The depth to water in the Santa Fe Group aquifer system varies considerably, ranging from less than 2 feet near the Rio Grande to more than 1000 feet in areas west of the river beneath the West Mesa (Bartolino and Cole, 2002). Recent investigations have indicated that the predominant groundwater flow direction in the valley in the area of Albuquerque has historically been north to south (Plummer et al., 2001; Bexfield and Anderholm, 2002). A 1936 groundwater level map indicates that the Rio Grande was losing water into the aquifer between Corrales and Belen. This losing reach during predevelopment conditions was probably not due to groundwater withdrawls, but rather evapotranspiration from vegetation and/or long-term recharge of Santa Fe Group aquifer system (Bartolino and Cole, 2002). The most current groundwater level map of the entire MRG basin was developed by Tiedeman et al. (1998) and represents winter 1994-1995 conditions. This map shows a general north to south flow with well-defined cones of depression in the Albuquerque and Rio Rancho areas associated with the withdrawal of groundwater from large-capacity production wells. More information on groundwater conditions in the Albuquerque and surrounding basin can be found in McAda and Wasiolek (1988); Kernodle et al., 1995; Anderholm, 1997; Hawley and Kernodle, 2000; Sanford et al., 2001; and McAda and Barroll, 2002.

2.4.8 Shallow Groundwater and Irrigation Effects

The depth to groundwater is an important determinant of vegetation type and density (Section 3.3). The riverside drains and LFCC intercept shallow groundwater flow and seepage from the river, and therefore, affect the direction of flow and depth of the water table. As shown on the conceptual model of the inner valley (Fig. 2-5), the riverside drains are in direct connection with the shallow groundwater system; this connection tends to create a constant hydraulic head (or water level) and gradient between the river and the drains. The drains result in a nearly permanent lowering of the shallow water table.

Historically, waterlogged lands associated with poor drainage and wasteful irrigation practices were a problem near the Rio Grande (Section 2.4.2). Under early agricultural methods, excess water in the acequias was dumped onto the low-lying lands near the end of the acequia causing a localized rise in the water table (Wozniak, 1987). Aggradation of the river channel in the 1880s further aggravated this situation. It was not until the 1930s that irrigation systems within the basin began to be improved through the installation and/or improvement of diversions and canals and riverside drains and levees. These improvements ultimately led to the conveyance of excess irrigation waters back to the river and a lowering of the shallow, induced water table.

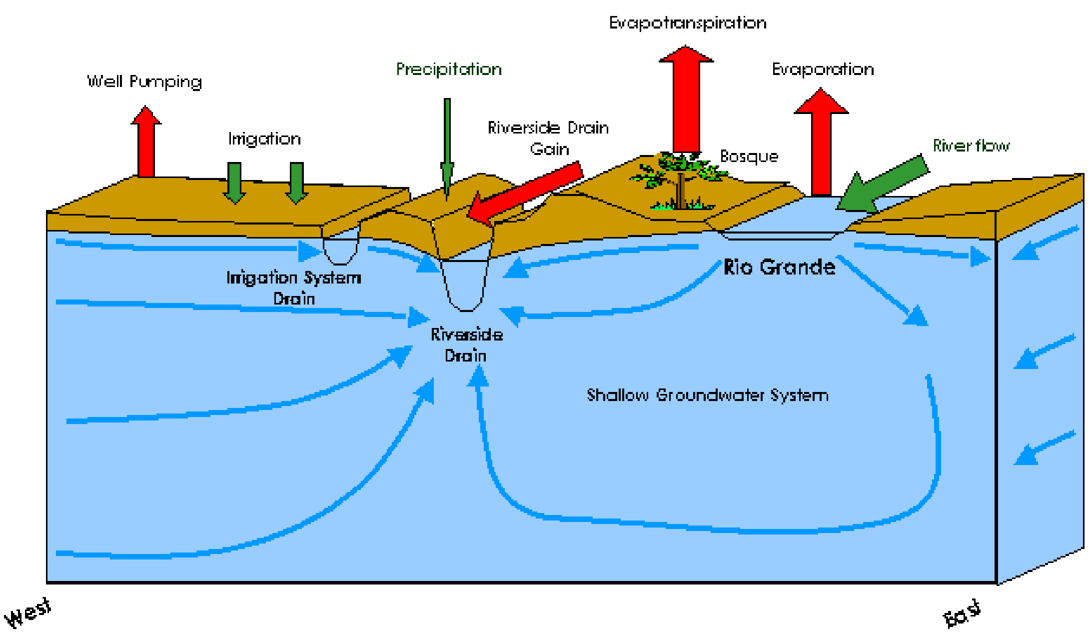

Figure 2-5: Conceptual model of the shallow hydrological cycle in the MRG

The inner valley of the Rio Grande currently contains an extensive system of irrigation canals, ditches, and drains, which have evolved from the early acequias. Acequias are the traditional irrigation ditches of the north, bringing water from the river and mountain tributaries to small farms along the river valley. Hundreds of acequias and ditches are still present and in use along the Rio Grande and its tributaries between the Colorado-New Mexico border and Cochiti Dam. The physical characteristics of the typical acequia system include a diversion dam and headgate, a main ditch, commonly called the acequia madre, lateral ditches leading from the main channel to irrigate individual parcels of land, and a wasteway ditch that returns surplus water back to the river. The channels are usually unlined, open and operate by gravity flow. The farms served by acequias range in size from less than one acre to more than 500 acres, with the majority being less than 20 acres (NMSEO, 1997).

Water has been and is currently diverted seasonally from the Rio Grande at a number of diversion dams and delivered to agricultural fields for the cultivation of crops. Additionally, there is a network of riverside drains that parallel the Rio Grande immediately outside the riverside levees. These drains and levees are generally located on both sides of the Rio Grande, except where bluffs adjoin the river. There are more than 180 miles of riverside drains and 160 miles of interior drains associated with about 128,000 acres of potentially irrigable lands that are overseen by the MRGCD between Cochiti Dam and Elephant Butte Reservoir.

Shallow groundwater flow outside of the riverside drains within the inner valley is largely controlled by the system of irrigation canals, ditches and drains that are present. The majority of the canals and acequias along the Rio Grande are unlined and constructed to allow water to be diverted onto irrigated fields. The base of the canals is generally above the adjacent land surface and above the top of the shallow water table in that area. Therefore, some of the water from the canals seep into and recharge the shallow groundwater system. Irrigation water applied to the fields that is not evaporated or transpired by crops drains to the shallow groundwater system. This water recharges the shallow groundwater system and flows toward the nearest downgradient drain or wasteway and returns to the Rio Grande (Fig. 2-5).

Shallow water level monitoring has been conducted as part of various investigations within the Middle Rio Grande basin (Bartolino and Niswonger, 1999; Eichhorst et al., 2002; Bowman et al., 2002). These investigations indicate that the depth to the shallow groundwater table within the MRG bosque ranges from several inches near the bank of the river to greater than 10 feet (essentially the elevation of the land surface and the top of the water level in the drain) near the riverside drains. Shallow groundwater data, collected as part of the Bosque Ecosystem Monitoring Program (BEMP) (Eichhorst et al., 2002), show that at most BEMP sites shallow groundwater is not responsive to river stage, and in general is only weakly correlated. Flows in the LFCC can be influenced by shallow groundwater. Analysis of groundwater data from 32 wells located along eight transects between San Acacia and San Marcial suggests that most of the flow within the LFCC during the winter is derived from seepage from the Rio Grande (Bowman et al., 2002).

2.5 Geomorphology

Fluvial geomorphology is the study of the dynamic interactions of water discharge, sediment load, regional and local geologic controls, and human interventions that affect the morphology of a river (Schumm, 1977). The cross sectional geometry (shape) and planform (pattern) of the Rio Grande's channel and its interaction with the adjacent floodplain are important determinants of habitat quality for the silvery minnow and flycatcher. The shape and gradient of the channel (water surface slope) affects the water velocity, degree of turbulence, and sediment transport capacity. The topography of the floodplain and height above the channel affects the ability of the river to connect to the floodplain at a particular discharge. Geomorphic processes work at different rates, with catastrophic events (e.g., low recurrence interval floods) causing large changes that may take significant periods of time to recover from under the influence of more routine flow conditions. Thus, understanding past and present geomorphological relationships and current trends are critical in the design and evaluation of restoration strategies.

Rivers are dynamic systems that adjust their nature in response to changes in flow and sediment regime. Like most rivers in alluvial valleys, the location, pattern, and cross-sectional profile of the Rio Grande changed episodically in response to natural variations in flow and sediment supply. Changes in the hydrology and sediment supply following construction and operation of the flood and sediment control facilities established in the 1950s and 1970s have been used to explain the modern channel morphology, including pattern, channel narrowing, channel incision, and changes in bed material composition (Crawford et al., 1993; Graf, 1994; Lagasse, 1994; Baird 1998 and 2001). The influence of water management operations on the geomorphology of the Rio Grande is undeniable; however, interpretations of cause and effect must also consider the degree of disequilibrium in the system prior to construction of the dams, rates of various processes under natural and regulated conditions, and the episodic nature of sediment dynamics. Interpretation of the Rio Grande geomorphology is complicated by the inherent complexity of the system and general paucity of data from both a temporal and spatial perspective.

2.5.1 Historical Geomorphology

Early descriptions of the Rio Grande come from journals and reports of the Spanish Colonial and later American Territorial expeditions (Scurlock, 1998). Common themes from these accounts include a river that was temporally and spatially dynamic, subject to seasonal and annual variations in flow, and episodically changed course. The width of the channel varied, with estimates at different times and places, and from different observers, ranging from 25 to more than 1,000 yards. Historical evidence can be found to indicate that the channel was either wider or narrower than the conditions that currently exist. Crawford et al. (1993), referring to the reaches below Cochiti, suggested that “generally the Middle Rio Grande was a braided, slightly sinuous aggrading river with a shifting sand substrate.” References to sloughs, ponds, bars and islands were not uncommon (Scurlock, 1998).

The earliest direct information on Rio Grande geomorphology below Cochiti comes from the 1917-1918 topographic maps published by the U.S. Reclamation Service (1922). It is unlikely that these surveys reflect pre-European conditions given the combined effects of hydrologic and watershed vegetation modifications in the late 1800s. Nonetheless, MEI (2002) interpreted the 1917-1918 surveys to indicate that the river was anastomosing, with vegetated islands between Cochiti and Angostura. A braided channel of varying width characterized the river from Angostura to Cañada Ancha. The channel narrowed and was confined where the river crosses the Belen-Socorro Uplift above the Rio Puerco confluence. A wide, braided channel with low sinuosity was evident from the Rio Puerco down to San Antonio. From San Antonio to San Marcial, the sinuosity of the channel increased. The average channel width generally increased downstream, with the widest areas associated with sediment contributions from the Rio Puerco and Rio Salado (MEI, 2002). Unfortunately, similar historical surveys were not performed for the Rio Chama and Rio Grande above Cochiti.

Historically, bars and islands (vegetated bars) were common features of the Rio Grande. Scurlock (1998) cites a report of the 1893 U.S. Census (Poore, 1894) indicating that several large islands occurred near Sandia Pueblo, “which rose about 6 feet above the level of the river and were covered with cottonwood groves. The uppermost island was estimated to be 700 acres.” In another report around 1850, a U.S. Army doctor described the Rio Grande near Lemitar and Cochiti as “…a rapid stream, about 120 or 200 feet wide, dividing off, so as to make many islands, the water is muddy and reddish, near the color of the Red River” (Ames, 1943:20 in Scurlock, 1998). Numerous point, alternate, and medial bars are shown on the 1917-1918 survey maps, including some that are vegetated (Figs. 2-6 and 2-7).

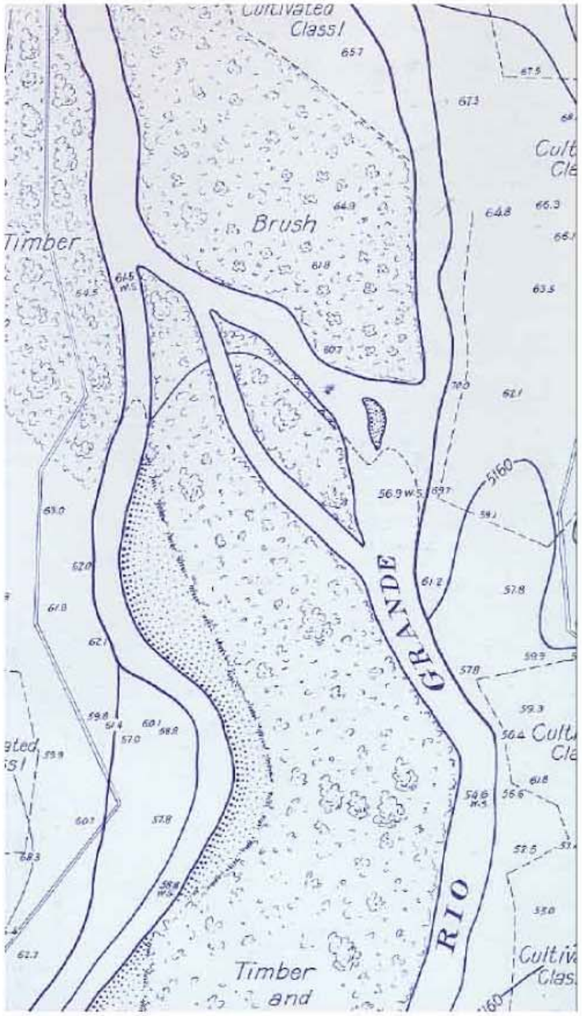

Figure 2-6a: Rio Grande between Isleta and Belen with mid-channel and lateral (1917/1918)

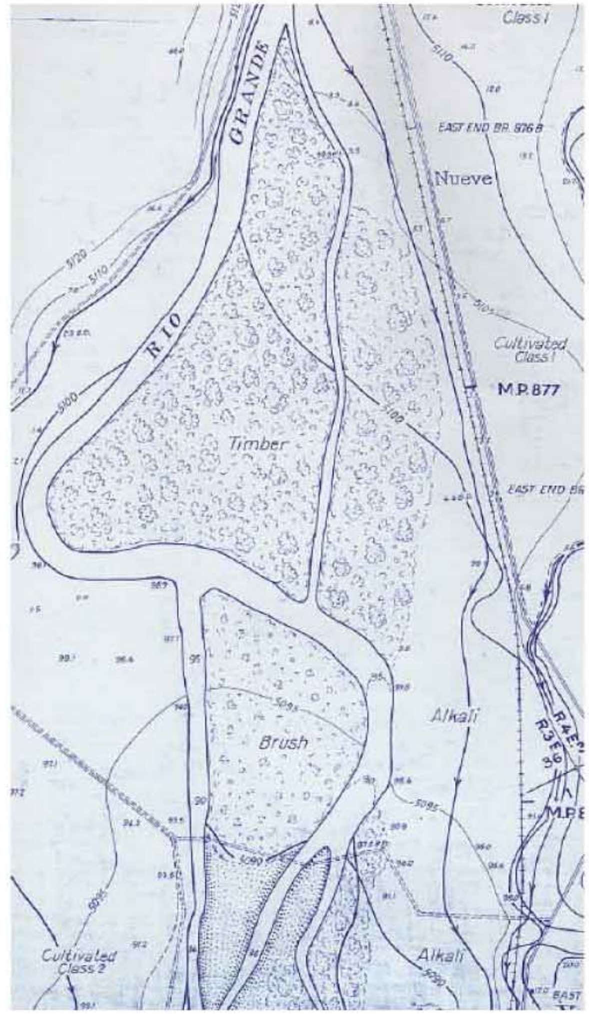

Figure 2-6b: Rio Grande between San Felipe and Angostura showing an anastomosed planform with one dominant channel (1917/1918)

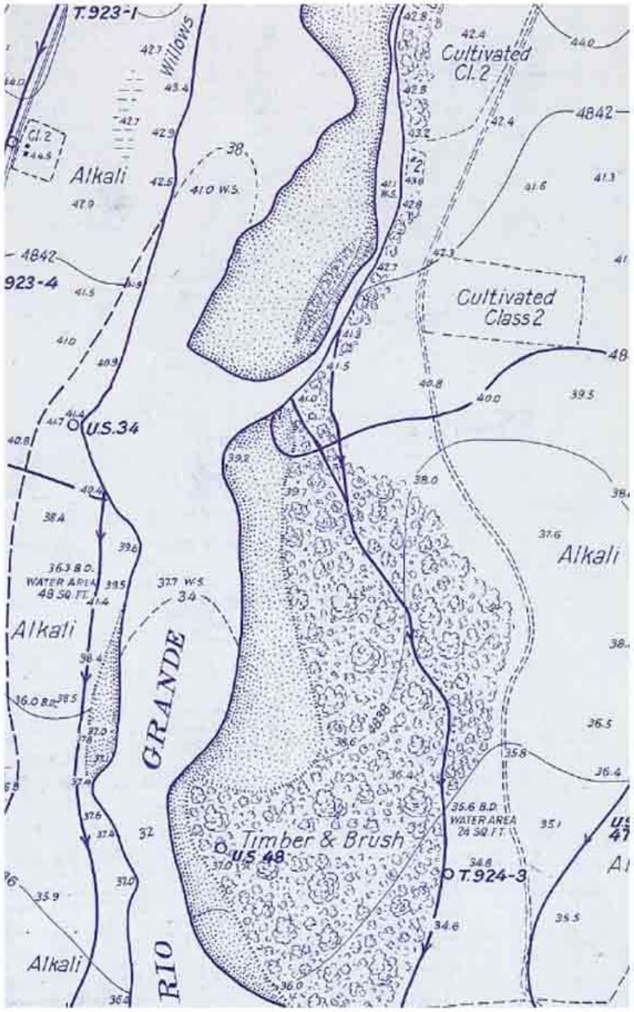

Figure 2-7a: Rio Grande between Isleta and Belen with mid-channel and lateral bars (1917/1918)

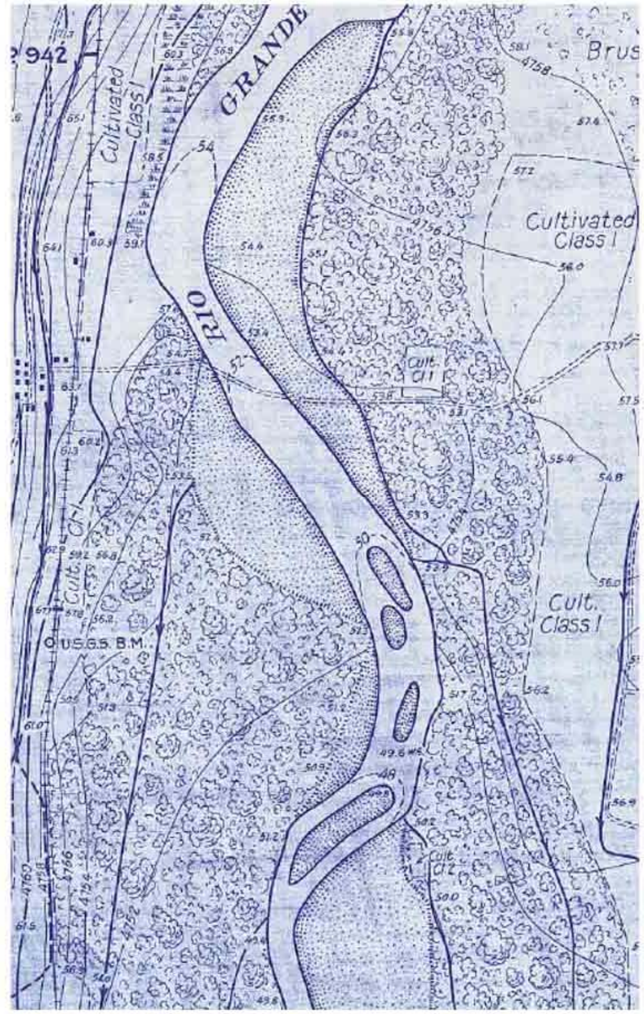

Figure 2-7b: Rio Grande between Belen and Cañada Ancha showing a variable-width, braided channel with mid-channel and lateral bars (1917/1918)

2.5.2 Current Geomorphology

Geomorphic conditions along the Rio Grande from Cochiti Dam to Elephant Butte Reservoir are described by MEI (2002). They concluded that realignment and channelization projects resulted in the conversion of the Rio Grande from a multi-channeled, multi-thalweg river through much of its length into a single channel. In general, the river is developing a low-sinuosity meandering channel within the levees associated with stabilization of bars and reduction in the magnitude of peak flows. About 60 percent of the river has a braided channel pattern under base-flow conditions between Cochiti Dam and Elephant Butte Reservoir. From Cochiti to near Isleta, many of the alternate bars have become attached to the banks and are being stabilized by vegetation.

Jetty-jack fields and other forms of bank protection currently limit lateral migration of the river and have eliminated a significant source of sediment, which may be contributing to the local channel incision (degradation) observed in the MRG. MEI (2002) indicated that localized bank erosion was insufficient to cause major changes in channel patterns. Furthermore, establishment of native and non-native vegetation has effectively stabilized much of the river, especially downstream of San Antonio, where the river has not incised below the rooting depth of the plants

2.5.2.1 Channel Width

Along the entire MRG, the mean channel width decreased by 24 to 52 percent between 1917-1918 and 1972 (MEI, 2002). Channel narrowing started before Cochiti Dam became operational (Baird, 2001). Evidence of widespread channel narrowing in the MRG between 1972 and 1992 is lacking, except for the reach from San Acacia to Escondida (MEI, 2002). MEI (2002) concluded that narrowing of the channel was primarily a result of channelization designed to increase the efficiency of flow conveyance. Thus, MEI (2002) concluded that the post-Cochiti Dam hydrologic and sediment regime did not cause channel narrowing in the upper reaches. They related the channel narrowing in the San Acacia to Escondida reach to the combined effects of channel incision (associated with channelization) and establishment of vegetation in the former river channel associated with flow reductions during the operation of the LFCC. Decreased sediment loads from the tributary watersheds may also influence channel widths (C. Gorbach, Personnel Communication). Studies performed by Reclamation generally support the conclusion that channel width has stabilized since 1972 (Massong et al., 2002). However, vegetation encroachment on bars in the floodway has contributed to a reduction in the amount of unobstructed channel since 1992 (Dello Russo and Obrien, 2000; Makar and Strand, 2002; Sixta et al., 2003).

The establishment of vegetation on the bars has resulted in localized channel narrowing under the flow and maintenance regime that characterized the last 5 to 7 years. The vegetated medial bars and islands may or may not become permanent features of the main channel depending on the 1) magnitude, duration and recurrence interval of peak flows 2) the age, type, and density of vegetation and 3) the potential for channel bed incision.

2.5.2.2 Aggradation/Degradation

Channel degradation, or downcutting, represents the manifestation of a disequilibrium condition that occurs when stream power exceeds resisting power (Bull, 1990). Degradation/aggradation may occur in response to changes in flow velocities, sediment regime (supply and character), and stream gradient (uplift or grade controls). Modern stream terraces, cross-sectional profiles, and field observations indicate that degradation of the Rio Grande channel is widespread from Cochiti Dam to Escondida, although local areas of aggradation may occur associated with tributary sediment dynamics and flow impoundments (e.g., diversions dams) (Massong et al., 2002; MEI, 2002). In contrast, aggradation is the dominant mode below Escondida.

The causes for degradation vary locally but can generally be attributed to (1) reduced sediment supply associated with operation of Cochiti Reservoir, as well as partial watershed recovery; (2) above normal precipitation in the late 1900s; (3) flow augmentation associated with the SJCP and municipal discharges; (4) suspension of the LFCC operation; (5), uplift associated with the Socorro magma body; and (6) channel realignment, narrowing, and maintenance downstream of Escondida (Massong et al., 2002; MEI, 2002). Leopold el al. (1992) indicated that aggradation below San Marcial was related to residual sediment from the large floods in the early 1900s, increased diversions of irrigation water during 1880-1915, which decreased the natural flow of the river at San Marcial by about one-half, and the railroad embankments that functioned as a sediment dam/barrier. Base level control imposed by Elephant Butte Reservoir has often been suggested as contributing to the aggradation observed in the San Acacia reach. However, Leopold et al. (1992) presented data spanning 1896 to 1935 showing that aggradation rates within this reach were similar both before and after the closure of the dam in 1915. The question remains regarding whether potential upstream reservoir effects may have become apparent at San Marcial since 1935.

Channel degradation reduces the ability of the river to access its floodplain under a given flow regime. MEI (2002) concluded that upstream of Isleta, only minor overbank flooding occurs at a discharge of 5,700 cfs, which has a recurrence interval of about 2 years. Between Isleta and Belen, overbank flooding can be generated at flows on the order of 5,700 cfs. Between Bernardo and San Acacia, the channel capacity is higher than 5,700 cfs, and therefore, the frequency of overbank flows is lower. Between San Acacia and San Antonio, flows up to about 5,700 cfs produce some overbank flooding, whereas below San Antonio, extensive overbank flooding is caused by flows in the same range (MEI, 2002).

Natural and artificial grade controls exist on the Rio Grande associated with near surface exposure of more resistant rock, coarse sediment from arroyos, and irrigation diversion structures. These features locally control the potential for degradation and aggradation (Lagasse, 1980). In other areas the rate or potential for degradation is affected by armoring associated with the occurrence of coarse textured bed materials.

2.5.2.3 Bed Characteristics

Because sand-dominated beds are more easily deformed than gravel- or cobble-dominated beds, the nature of the bed materials affects the response of the channel under various flow regimes. Flood and sediment control reservoirs have had a major effect on sediment and river dynamics downstream of Cochiti Dam (Baird, 1998 and 2001). Suspended sediment loads have decreased relative to the periods before the construction of Abiquiu and Cochiti dams; however, the effects of the dams diminish downstream because of tributary contributions and in-channel sediment sources (MEI, 2002). Downstream from the Rio Puerco confluence, tributary and in-channel sediment sources reduce the potential effects of sediment reductions related to dam operations. In the post-Cochiti Dam period, average annual suspended-sediment concentrations decreased by about 99 percent at the Cochiti gage and by 70 percent at the San Marcial gage. Notably, during this period, the average annual suspended-sediment concentrations decreased nearly 55 percent at the Otowi gage upstream of Cochiti Reservoir (MEI, 2002). Also, as discussed in Section 2.3.6, there has been a notable decrease is sediment contributions from the Rio Puerco in recent years (Elliott et al., 1999).

Basin-wide watershed factors likely explain the reduction of sediment loads at Otowi. Elliott et al. (1999) attributed the decline in sediment load to improved land management practices, reforestation, fire suppression, and the storage of sediment in arroyos that incised in the 1800s and have subsequently stabilized. Furthermore, it is likely that the relatively wet conditions that prevailed in the 1980s and 1990s have affected watershed conditions and sediment production and routing within the MRG basin. A return to drier conditions or severe drought may result in increased sediment production as upland vegetation cover decreases and runoff is accentuated. Although counter-intuitive, historical data indicate the relatively large peak flow events are not uncommon during drought periods.

The description of bed characteristics and interpretation of trends is complicated by the relative paucity of historical and current data. Nonetheless, there is fairly broad agreement that the reach immediately below Cochiti Dam has coarsened in response to sediment retention associated with the operation of the dam, and the bed is currently dominated by cobbles and gravels. The channel bed materials tend to become finer in texture downstream ranging from sand-dominated conditions in the reaches below Bernalillo to a clay and silt substrate near Elephant Butte Reservoir delta (Massong et al., 2002; MEI, 2002). Zones with higher proportions of gravels occur locally, and especially in association with higher-gradient tributaries and below diversion structures.

2.6 Vegetation

Upland vegetation in the MRG basin ranges from coniferous spruce-fir and mixed conifer forests at the higher elevations to desert scrub and desert grasslands at the lower elevations. Pinyon-juniper woodlands occur extensively in the middle-elevation range (Brown, 1982; Gottfried et al, 1995; Loftin et al., 1995; Dick-Peddie, 1993). One of the most extensive and continuous riparian forests (bosque) in the southwestern United States occurs in the broad valleys along the Rio Grande in New Mexico (Hink and Ohmart, 1984). Crawford et al. (1993) suggest that the Rio Grande bosque has existed in some form for more than 2 million years. These prehistoric stands were probably dominated by cottonwood with subdominant allies including birch (*Betula* spp.), cherry (*Prunus* spp.), boxelder (*Acer negundo*), western soapberry (*Sapindus drummondii*), velvet ash (*Fraxinus velutina*), netleaf hackberry (*Celtis reticulata*), and Arizona sycamore (*Platanus wrightii*) (Crawford et al., 1993; Hink and Ohmart, 1984).

2.6.1 Historical Riparian Vegetation

By the time the initial biological and hydrological surveys were conducted in the Middle Rio Grande (Watson, 1912; Bureau of Soils, 1914; U.S. Reclamation Service, 1922), the basin had undergone dramatic changes due to agricultural development. Grazing and timber harvesting in the uplands, compounded by the severe 1899 to 1904 drought, reduced upland plant cover and contributed to accelerated erosion in the tributary watersheds, which led to aggradation in portions of the MRG. Upstream diversions, primarily in the San Luis basin of Colorado, decreased base flows. Moreover, many of the local irrigation systems in the MRG were open-ended, dumping water on the alluvial terraces at the end of ditches rather than returning it to the river. The poor management of the irrigation ditches combined with the reduction in flow and increased sediment load to raise the groundwater table (Wozniak, 1998). In turn, salinity and drainage problems in the valley became increasingly severe and widespread (Watson, 1912; Bureau of Soils, 1914; U.S. Reclamation Service, 1922).

Up until the mid to late 1800s, the anastomosed and braided river had a floodplain 1 to 4 miles wide with a mosaic of cottonwood/willow forest, locally abundant saltgrass meadows, and occasional marshes. Islands may have been vegetated with grass or trees. Crawford et al., 1993 indicated that the presence and depth to ground water determined the composition of plant communities and their distribution within the floodplain. Historically, establishment of new stands was associated with episodic disturbances caused by floods. In general, there were three dominant plant communities in the MRG floodplain: cottonwood bosque with a willow understory, grass meadows (alkali or wet), and wetlands (Crawford et al., 1993). The cottonwood/willow bosque occurred in areas with a shallow water table that fluctuated seasonally but rarely stayed near the surface for extended periods (Section 3.3). Areas where groundwater was at or just below the surface accumulated significant quantities of salts on the surface through evapoconcentration, affecting the composition of the plant community. Saltgrass and other salt-tolerant plants typically would dominate the wet, saline-alkali meadows. Flow-through wetlands, containing cattails, sedges, rushes, and emergent vegetation, likely occurred in sloughs and oxbows (Crawford et al., 1993).

Early Spanish accounts recognized the occurrence of wetlands. In 1918, areas with water tables very near the surface occupied about 6,200 acres between Cochiti and San Marcial, representing nearly 15 percent of the area surveyed (U.S. Reclamation Service, 1922). Because the water table was artificially elevated by that time, wetlands would probably have occupied less area under proper irrigation management.

2.6.2 Existing Riparian Vegetation

The amount of area categorized as riparian forest in the MRG has not appreciably changed over about the last century, although the distribution and composition of the stands are different (Crawford et al., 1993). The major changes in composition are associated with the introduction of invasive exotic tree species and a reduction in the age structure of the overstory dominants. Introduced for erosion control, windbreaks, and ornamental uses in the late 1800s, saltcedar (*Tamarix* spp.) rapidly expanded in southern portions of the MRG (MacDonald, 1955). North of Albuquerque and into the Rio Chama, Russian olive (*Elaeagnus angustifolia)* has outpaced saltcedar as the dominant invader (Dick-Peddie, 1993). Four scouring floods between 1929 and 1942 opened large areas that were colonized by saltcedar and Russian olive, particularly south of Bernardo. Concurrently, the MRGCD began to construct drainage and irrigation facilities in the 1930s, dropping groundwater tables and making more land available for agricultural production. As the lands were drained, they were either converted to agricultural fields or were colonized by cottonwood, willow, saltcedar and Russian olive.

2.7 Depletions

River and riparian restoration projects are proposed in the MRG with the intent of protecting silvery minnow and flycatcher habitats. Restoration practices that are broadly considered appropriate for the Rio Grande include manipulation of the flow regime, physical modifications of the channel/floodplain, and vegetation management. Water use requirements of the restoration activities vary depending on the nature, extent, and location of the project. Because restoration activities can result in gains or losses of water and have the potential to change the river hydrograph, characterizing the water requirements of restoration activities is important in order to maximize the benefits to the listed species, while still maintaining New Mexico's obligations relative to the Compact and providing water to valid water right holders throughout the MRG.

Water in the MRG basin is fully appropriated. Current estimates indicate that on average New Mexico has about a 50 percent chance of meeting the Compact delivery requirements in any year (SSP&A, 2000). In the absence of exercising a water right based upon prior appropriation for beneficial use, the New Mexico Office of the State Engineer requires that any citizen or political subdivision of the state that pumps groundwater or diverts surface water that depletes native Rio Grande water must offset such depletions. In effect, any new use of water in the MRG requires that an existing use be retired.

2.7.1 Rio Grande Compact and Depletions

The goal of the Compact was to equitably apportion the waters of the Rio Grande among Colorado, New Mexico, and Texas based on conditions that existed in 1929. Briefly, the Compact requires Colorado to deliver about one third of the flow of the Rio Grande originating in Colorado to New Mexico in average years, about one fourth of the flow in dry years, and about two thirds of the flow in wet years. Native Rio Grande flows at the Otowi gage corrected for upstream storage of water define delivery requirements from New Mexico to Texas. About 60 percent of the native Rio Grande inflow past Otowi must be delivered to Texas in dry years and over 80 percent in wet years. New Mexico's total allocation of native inflow at Otowi is capped at 405,000 AFY (Fig. 2-8). However, all of the highly variable tributary inflows below the Otowi gage can be used in the MRG. New Mexico's deliveries are measured as releases below Elephant Butte Reservoir, plus the net change in storage in the reservoir. SJCP water flowing past Otowi is excluded from Compact accounting and must be used consumptively within New Mexico.

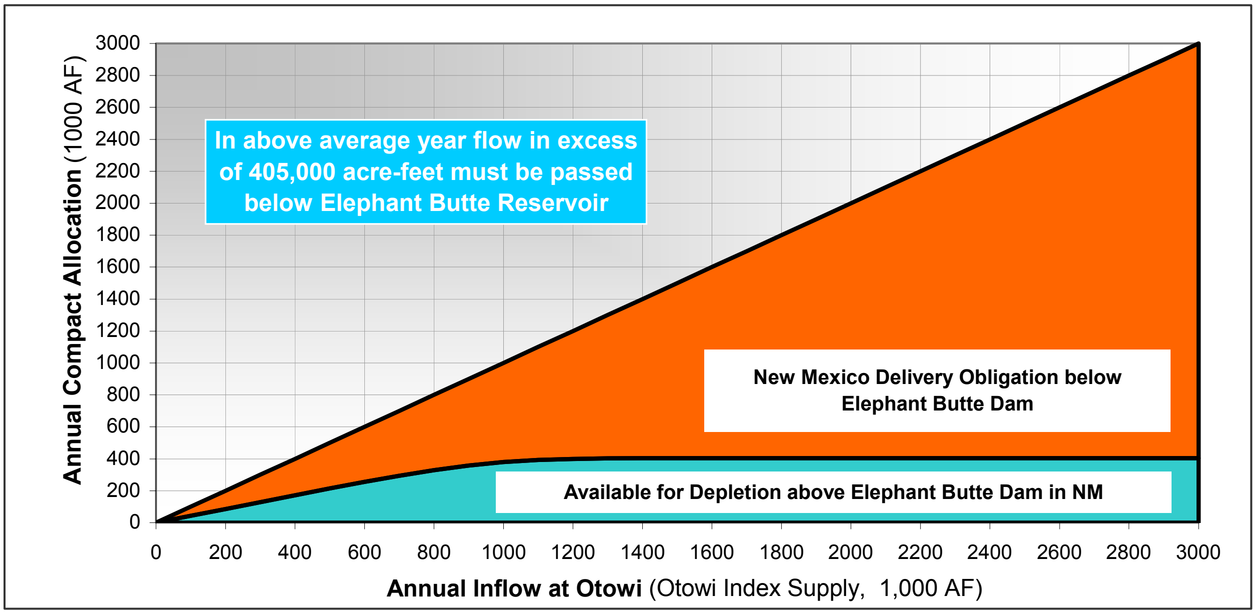

Figure 2-8: Annual Compact Allocation
(SSP&A, 2000)

In the context of the Compact, depletions are defined as irrecoverable evaporation and transpiration losses from agricultural and riparian vegetation, open water, bare channel sediments, and municipal and industrial use. Consumptive use refers to losses of water from a hydrologic system over a specified period through evaporation from soils and transpiration from plants, including water that is used to build plant tissue. In general, the magnitude of annual water loss from different surfaces occurs in the following order: pan evaporation > open water ≥ bare saturated soil > riparian vegetation > upland vegetation > dry soil. Because surface water and groundwater are connected in the MRG, seepage, transference losses, agricultural leaching, and groundwater recharge are not considered depletions; however they may affect annual Compact accounting. Similarly, groundwater pumping and increased transference efficiencies do not represent net gains in basin water supply. Crop and riparian ET account for the largest depletions in the MRG followed by reservoir evaporation and urban depletions (Fig. 2-9).

2.7.2 Reservoir and Open Water Depletions

Open water evaporation represents a significant source of water loss in the MRG. The evaporative demand in this region is high with average pan evaporation rates in the range of about 5 feet per year (1,200 to 1,525 mm/y) at El Vado to almost 10 feet per year (3,000 mm/y) at Elephant Butte Reservoir. Evaporation from Cochiti (5,000 to 20,000 AFY) and Elephant Butte (50,000 to 250,000 AFY) Reservoirs accounts for about 19 percent of the average annual water loss in the MRG (SSP&A, 2000).

Reservoir evaporation is expected to range from about 60 to 85 percent of pan evaporation depending on the size of the water body and other factors. Smaller water bodies tend to have evaporation rates closer to pan evaporation rates.

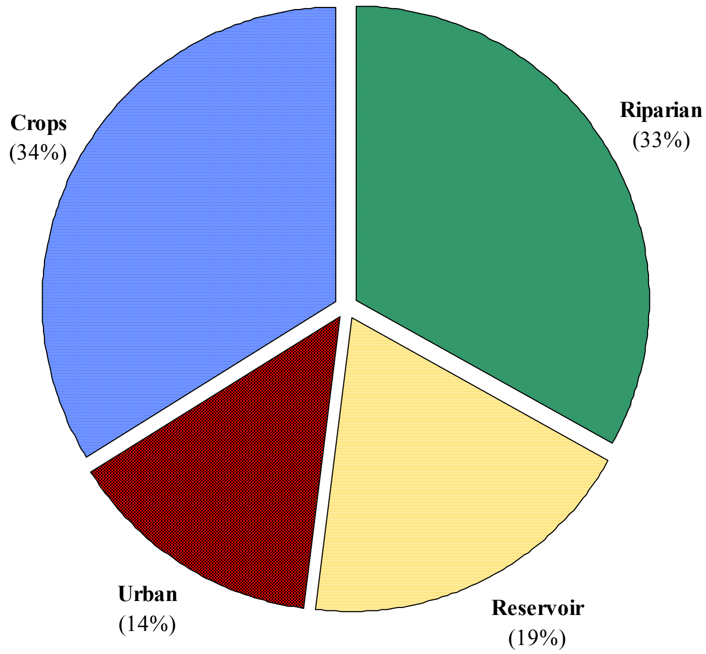

Figure 2-9: Mean total depletions in the MRG under present land use and groundwater development conditions
(SSP&A, 2000)

Evaporation from the river, bare channel sediments, canals, and drains are also important components of the regional water balance. For instance, the active (bankfull) channel of the Rio Grande from Cochiti to the upper end of Elephant Butte Reservoir is estimated to occupy about 10,000 acres, excluding drains and canals. For comparison, the July 2002 pool surface of Elephant Butte Reservoir was about 10,000 acres. At maximum capacity Elephant Butte Reservoir covers about 36,500 acres. Evaporation from wetted channels, canals, ditches, and drains is expected to occur at rates similar to those discussed above for open water bodies, although shading may reduce the rate in some instances. Thus, during low storage years, channel and conveyance system depletions may rival losses from the reservoirs.

The rate of evaporation from saturated soils may equal open water evaporation rates during periods of low to moderate evaporative demand (Hillel, 1998). Soil hydraulic properties and water table depth control the potential rate of evaporation from bare channel sediments. Thus, evaporation from exposed channel sediments may be significant during low flow periods and this effect should be considered for restoration activities that involve channel widening. Unlike vegetated areas where dormancy and shading reduce evaporation rates during the non-growing season, evaporation from open water and bare saturated soils occurs at the maximum climatically determined rate throughout the year (i.e., as affected by temperature, heat transfer, and wind speed). Water loss from the river channel, canals, and drains is currently included in the depletion estimates for the riparian zone (SSP&A, 2000).

2.7.3 Riparian Vegetation Depletions

Consumptive use associated with the riparian zone accounts for about a third of the average annual depletions in the MRG (SSP&A, 2000). Because the estimated riparian zone depletions include open water evaporation from the channel, the amount of water lost strictly to ET is unknown. The potential for creating habitat for the listed species while reducing depletions through vegetation manipulations is the driving force behind understanding riparian depletions. Consumptive use of agricultural crops has been the subject of significant research and forms the fundamental basis for understanding plant/soil water relations (Doorenbos, et al., 1992; Hillel, 1998). Alternatively, riparian vegetation ET is not well quantified from a comprehensive perspective, with past research focused mainly on saltcedar, rather than the broad range of plants in riparian communities (Moore et al., 2000). Extrapolation of ET rates is problematic, because ET depends on complex interactions between plants, the soil, and atmosphere (Hillel, 1998), and their estimation is method dependant (WMO, 1971). Eddy covariance methods are thought to provide the most accurate estimates of ET and only a few studies have used this approach for riparian vegetation in the MRG.

Saltcedar stands are generally considered to use more water per unit area than native riparian stands (Gatewood et al., 1950; King and Bawazir, 2000). Sala et al. (1996) indicated that saltcedar ET rates were no greater than native phreatophytes on a leaf area basis, but saltcedar has the ability to form stands with higher total leaf areas than native phreatophytes, resulting in higher overall water loss. Recent studies in the MRG using eddy covariance methods support these general relationships. King and Bawazir (2000) reported that annual ET of 4.4 feet (1330 mm) from a dense saltcedar stand compared to 3.0 feet (904 mm) for a sparse cottonwood stand at Bosque del Apache National Wildlife Refuge. Cleverly et al. (2002) reported that annual ET from saltcedar stands measured in 1999 varied from about 4.2 feet (1220 mm) at a flooded site to 2.4 feet (740 mm) for a non-flooded site. Coonrod and McDonnell (2001) indicated that annual ET measured from cottonwood stands in 2000 varied from about 2.4 feet (720 mm) at a flooded site to 3.0 feet (930 mm) at a non-flooded site. During the same measurement period, annual ET from saltcedar stands varied from 2.6 feet (780 mm) at a flooded site to 2.0 feet (600 mm) at a non-flooded site (Coonrod and McDonnell, 2001).

The variations in measured ET for saltcedar and native stands indicate that local differences in stand characteristics, soils, ambient climate, depth to water table, and flooding complicate the extrapolation of ET data over time and space. Furthermore, these data suggest that conversion of saltcedar stands to native riparian communities may not always reduce depletions.

Reclamation developed the ET Toolbox to estimate daily water losses associated with crop and riparian ET and open water evaporation within specified river reaches (Brower, et al., 2001). ET Toolbox makes estimates of water loss for various land, vegetation, and water components on a 4-km grid. Daily ET is estimated using a modified Penman equation corrected with experimentally derived crop coefficients. The output from ET Toolbox is used to support the river modeling and water accounting system (RiverWare) used by the Upper Rio Grande Water Operations Model (URGWOM). The application of ET Toolbox to evaluate restoration activities is limited by the grid size, but the basic information used to develop the model is considered important to understanding depletions in the MRG. URGWOM is a numerical computer model being developed to simulate water storage and delivery operations in the Rio Grande from its headwaters in Colorado to below Caballo Dam in New Mexico and for flood control modeling from Caballo Dam to Fort Quitman, Texas. The model is used to aid in flood control operations, water accounting, and evaluating water operations alternatives.

Accurate determinations of riparian depletions in the MRG would require site-specific characterization of soil, topographic, flood, groundwater, and vegetation conditions coupled with a quantitative understanding of the response of vegetation to variations in these physical conditions. However, until ET relationships are more accurately defined on a site-specific and Program basis, conservatively-biased consumptive use values can be used to estimate restoration-related effects on basin depletions. Research continues on ET rates in different riparian communities and evaporation rates from open water and bare soils to better understand water loss along the MRG. As information on water loss via ET is improved, estimates for water use and plans for limiting depletions within the Program area will be updated.

3.0 HABITAT NEEDS AND ECOLOGICAL RELATIONSHIPS

The development of a habitat restoration plan for endangered species purposes requires a clear understanding of the biology and ecology of the listed species and their habitat. The intent of this section is to provide information on the biology, ecology, and habitat needs of the silvery minnow (Section 3.1) and flycatcher (Section 3.2). The section concludes with an introduction to the biological and ecological relationships for the riparian plant community.

3.1 Rio Grande Silvery Minnow

The entire wild population of the silvery minnow is restricted to portions of the MRG. Our understanding of silvery minnow habitat comes from field observations under contemporary conditions and comparisons to related species. Research is ongoing, but many aspects of the biology and reproductive ecology of the silvery minnow are poorly understood. The BA cautions "that all investigations of life history and ecology of the silvery minnow have taken place within the species' contemporary range, an environment that has been dramatically altered over historic times. Observations from such investigations can easily lead to a misunderstanding of the species' habitat preferences and needs" (BOR and COE, 2003, p. 14). Consequently, much of the discussions regarding the life history, ecological characterization, and habitat relationships presented herein for the silvery minnow should be considered as working hypotheses.

3.1.2 Status and Distribution

The silvery minnow is a Federal and State (New Mexico and Texas) listed endangered species (FWS, 1994; NMGF, 1996; Texas Parks and Recreation, 2003). Historically, it was one of the most common fish in much of the Rio Grande (FWS, 1994). Silvery minnows ranged from near the Gulf of Mexico to Española, New Mexico on the main stem of the Rio Grande and up to Abiquiu, New Mexico on the Rio Chama (Bestgen and Platania, 1991). The silvery minnow also occurred in the Pecos River from Santa Rosa, New Mexico south to the confluence with the Rio Grande. Currently, silvery minnows inhabit approximately 5 percent of their historical range, with the entire wild population occurring in the Rio Grande between Cochiti Dam and the Elephant Butte Reservoir delta (FWS, 2003b).

Six native minnow species, including the silvery minnow, and three introduced minnow species currently persist in the MRG (Table 3-1; Propst, 1999). Fourteen of the 27 native fish species that once occupied the MRG have become extinct or been extirpated (BOR and COE, 2003). Today, over 30 fish species inhabit the MRG, with at least 22 non-native (i.e., exotic) fish species having been collected (Table 3-1).

Table 3-1: Historical and contemporary fish species in the MRG
(BOR and COE, 2003; Sublette et al., 1990)

FAMILY AND SPECIES*	COMMON NAME	HISTORICAL	CURRENT
CIPENSERIDAE (sturgeons)			
Scaphirhynchus platorynchus (n)	shovelnose sturgeon	*X*	-
ANGUILLIDAE (freshwater eels)			
Anguilla rostrata (n)	American eel	*X*	-
CATOSTOMIDAE (suckers)			
Carpiodes carpio (n)	river carpsucker	*X*	*X*
Catostomus (Catostomus) commersoni (e)	white sucker	-	*X*
Catostomus (Pantosteus) plebeius (n)	Rio Grande sucker	*X*	*X*
Cycleptus □andora□s (n)	blue sucker	*X*	-
Ictiobus bubalus (n)	smallmouth buffalo	*X*	*X*
Moxostoma congestum (n)	gray redhorse	*X*	-
CENTRARCHIDAE (sunfishes)			
Lepomis (Chaenobryttus) cyanellus (e)	green sunfish	-	*X*
Lepomis macrochirus (n)	bluegill	*X*	*X*
Micropterus dolomieu (e)	smallmouth bass	-	*X*
Micropterus punctulatus (e)	spotted bass	-	*X*
Micropterus salmoides salmoides (e)	largemouth bass	-	*X*
Pomoxis annularis (e)	white crappie	-	*X*
Pomoxis nigromaculatus (e)	black crappie	-	*X*
CHARACIDAE (characins)			
Astyanax mexicanus (n)	Mexican tetra	*X*	-
CLUPEIDAE (herrings)			
Dorosoma cepedianum (n)	gizzard shad	*X*	*X*
Dorosoma petenense (e)	threadfin shad		
CYPRINIDAE (minnows)			
Carassius auratus (e)	goldfish	-	*X*
Cyprinella lutrensis (n)	red shiner	*X*	*X*
Cyprinus carpio (e)	common carp	-	*X*
Dionda episcopa (n)	roundnose minnow	*X*	-
Gila □andora (n/E)	Rio Grande chub	*X*	*X*
Hybognathus amarus (n)	Rio Grande silvery minnow	*X*	*X*
Macrhybopsis aestivalis aestivalis (n)	speckled chub	*X*	-
Notemigonus crysoleucas (e)	golden shiner	-	*X*
Notropis jemezanus (n/E)	Rio Grande shiner	*X*	-
Notropis orca (n/E)	phantom shiner	*X*	-
Notropis simus simus (n/E)	bluntnose shiner	*X*	-
Pimephales promelas (n)	fathead minnow	*X*	*X*
Platygobio gracilis (n)	flathead chub	*X*	*X*
Rhinichthys cataractae (n)	longnose dace	*X*	*X*
ESOCIDAE (pikes)			
Esox lucius (n)	northern pike	-	*X*

(*e*) = exotic, (*n*) = native, and (*n/E*) = native extinct species in the Rio Grande of New Mexico. All names according to AFS (1991).

Table 3-1 (continued): Historical and contemporary fish species in the MRG

FAMILY AND SPECIES*	COMMON NAME	HISTORICAL	CURRENT
ICTALURIDAE (bullhead catfish)			
Ameiurus melas (e)	black bullhead	-	X
Ameiurus natalis (e)	yellow bullhead	-	X
Ictalurus punctatus (e)	channel catfish	-	X
Ictalurus furcatus (n)	blue catfish	X	-
Pylodictis olivaris (n)	flathead catfish	X	X
LEPISOSTEIDAE (gars)			
Lepisosteus osseus (n)	longnose gar	X	-
PERCICHTHYIDAE (temperate basses)			
Morone chrysops (e)	white bass	-	X
Morone saxatilis (e)	striped bass	-	X
PERCIDAE (perches)			
Perca flavescens (e)	yellow perch	-	X
Stizostedion vitruem (e)	walleye	-	X
POECILIIDAE (livebearers)			
Gambusia affinis (n)	western mosquitofish	X	X
Poecilia latipinna (e)	sailfin molly	-	X
SALMONIDAE (trouts)			
Oncorhynchus mykiss (e)	rainbow trout	-	X
Oncorhynchus mykiss. X clarki hybrid (e)	cutthroat-rainbow hybrids/cutbows	-	X
Oncorhynchus clarki virginalis (n)	Rio Grande cutthroat trout	X	-
Salmo trutta (e)	brown trout	-	X
SCIAENIDAE (drums)			
Aplodinotus grunniens (n)	*freshwater drum*	X	-

(*e*) = exotic, (*n*) = native, and (*n/E*) = native extinct species in the Rio Grande of New Mexico. All names according to AFS (1991).

3.1.2 Contemporary Environmental Stresses

Declines in the silvery minnow population are broadly attributed to "dewatering, channelization and regulation of river flow to provide water for irrigation; diminished water quality caused by municipal, industrial, and agricultural discharges; and competition with or predation by non-native species" (FWS, 1994). However, the relative contributions of each these factors to the overall decline of the silvery minnow are difficult to gauge. These factors are discussed below.

3.1.2.1 Channel Dewatering

As discussed in Section 2.4.4, the river rarely dries up between Cochiti Dam and Isleta. However, substantial portions of the Isleta and San Acacia Reaches dried during the irrigation season in 2002 and 2003. This periodic and sometimes extensive drying of the channel in the Isleta and San Acacia reaches makes conservation and recovery efforts for the silvery minnow very difficult. River drying negatively affects the silvery minnow by reducing the amount and quality of habitat available for the populations. This reduction in habitat due to river drying can result in a decrease in silvery minnow fitness, as well as direct mortalities. The frequency, duration, and rate of river drying are factors that influence silvery minnow survival.

3.1.2.2 Habitat Fragmentation

Reservoirs and diversion dams disrupt the longitudinal connectivity of the river (BOR and COE, 2003), affect channel morphology and sediment dynamics, and alter natural cycles of flow and water temperature (e.g., Ward and Stanford, 1983). Disruption in longitudinal connectivity caused by irrigation diversions, which prevents upstream movement of fish, has been suggested as a factor contributing to the decline of the silvery minnow (FWS, 1994; Platania and Dudley, 2003). Similarly, Cochiti Dam is postulated to have prevented the dispersal of eggs downstream (BOR and COE, 2003). The persistence of the silvery minnow in the MRG is inferred to have depended on a stable population in the Otowi to Bernalillo transition zone (BOR and COE, 2003). Silvery minnow and other native pelagic broadcast spawning fish that historically inhabited these reaches functioned as brood stock and "factored prominently in the probability of colonization of empty downstream habitat patches (e.g., recently rewetted river channel)" (BOR and COE, 2003).

Although the silvery minnow has coexisted with irrigation diversions for nearly 70 years, the impact of these structures on their population dynamics is uncertain. Restricting upstream movement of fish may lead to a loss of the genetic diversity. The potential impacts of habitat fragmentation on genetic diversity of the silvery minnow have yet to be modeled.

The impact of irrigation structures on the upstream movement of silvery minnows to restock natal areas also remains unresolved. Laboratory studies indicate that silvery minnows have strong swimming abilities and could navigate fish passageways (Bestgen et al., 2003). Recent field releases of laboratory cultured silvery minnows indicate that at a least small fraction of these fish are capable of swimming upstream (Platania et al. 2003). However, as discussed below (Section 3.1.4.5), it is uncertain whether silvery minnows have the behavioral disposition to move upstream in significant numbers or whether they would use fish passage structures. The construction of fish passage at diversions may allow individuals to swim upstream prior to spawning, thereby reducing net downstream displacement of the population and could reduce the potential for genetic simplification.

3.1.2.3 Channelization and Flow Regulation

Changes in channel configuration associated with channel straightening, maintenance, and grade control affect the distribution of flows and aquatic habitat. Historically, diverse aquatic habitats resulted from eddies and other flow retarding features associated with intermittent side channels, active and vegetated bars and islands, shoreline embayments, sloughs, and bank irregularities along both the channel and island shorelines; cobbles, and boulders in some reaches; and accumulations of large woody materials. Combined, these features provided cover habitat for fish not only during high flows but also during lower flows. During large floods, overbank flows and channel avulsions dispersed the flow and probably retarded the downstream displacement of eggs. Unfortunately, channelization and maintenance operations have promoted a configuration where in-channel habitat diversity has decreased.

Flow regulation has been inferred to have impacted the silvery minnow (FWS, 1994), although the precise mechanism has not been fully elucidated. As discussed in Section 2.4.4, the operation of upstream reservoirs has resulted in the attenuation of the peak hydrograph and extension of the low flow period. Thus, flows in the river have been more consistent than in times prior to the operation of the reservoirs.

3.1.2.4 Water Quality

The listing for the silvery minnow states that degraded "water quality caused by municipal, industrial, and agricultural discharges" poses a threat to the silvery minnow (FWS, 1994). Elevated water temperatures and depletion of dissolved oxygen in pools associated with the drying river contribute to the mortality of silvery minnows in the Rio Grande (FWS, 2003b). Silvery minnows are typically associated with water temperatures in the approximate range of 35°F (1°C) to 85°F (30°C; FWS, 2003b).

Conclusive evidence is lacking that degraded water quality is adversely affecting silvery minnow in the MRG. Several studies have concluded that water quality conditions in the Rio Grande have not shown acute toxicity, but evidence is lacking on whether chronically toxic water quality conditions may sometimes affect silvery minnows. For example, Parsons Engineering (2000) evaluated water quality data from runoff samples collected from six locations in the City of Albuquerque from May to October 1992; concluding that the pollutants evaluated in runoff did not exceed acute water quality standards. Parsons Engineering (2000) also reported that a single runoff event from one site in 1999 did not produce acute toxicity responses in the animals tested. In 2001, based on available data and their sampling of the MRG, the New Mexico Environment Department (NMED) concluded water quality in the MRG was not impairing aquatic life (NMED, 2001). Buhl (2002) conducted laboratory toxicity tests on silvery minnows using various concentrations of aluminum, ammonia, arsenic, chlorine, copper, and nitrate. He concluded, "these chemicals individually or combined as environmentally relevant concentrations do not pose an acute hazard to populations of Rio Grande silvery minnow and fathead minnow. However the margins of difference between the acutely toxic concentrations of copper and ammonia in the mixture and those measured in the Rio Grande indicate that this mixture may pose a chronic hazard to both species." Therefore, while completed water quality evaluations have not demonstrated short-term acutely toxic conditions, further investigations are required to assess the potentials for long-term chronic toxicity effects on silvery minnow growth and reproduction, and to assess potential indirect effects of water quality on primary production (food chain effects).

Levings et al. (1998) summarized the results of a 1992 to 1995 USGS water quality study that assessed conditions at 17 sites along the Rio Grande from Del Norte, Colorado to downstream of El Paso, Texas. While one or more pesticides and their metabolites were detected in bed sediments at most of the sampling sites and in some samples of whole-body fish tissue, most concentrations were at or only slightly above the method detection limits. Overall, clear relationships were lacking to link these study results to potential adverse impacts to silvery minnows. Nonetheless, concerns exist regarding the potential effects on aquatic organisms from pharmaceutical drugs, antibiotics, synthetic hormones, and related chemicals discharged with treated wastewaters. The New Mexico Department of Health and the NMED have conducted surveillance surveys for selected pharmaceutical residues in the MRG. Caffeine and estrone were detected approximately 3 kilometers downstream from a municipal sewage effluent outfall (McQuillan and others 2001). Estrone was detected at 140 nanograms per liter, which is within the range that Jobling et al. (1998) attributed sexual disruption in wild fish. Additional investigation is required to define the role, if any; pharmaceutical residues may have on the population of silvery minnow.

The 2003 BO stipulates at Reasonable and Prudent Alternative (RPA) DD that, "With the increased emphasis and importance of the Angostura Reach for silvery minnow conservation, it is imperative that the addition of treated wastewater to the river provides water quality conditions protective of silvery minnow. The protective concentration of total residual chlorine (chlorine) for silvery minnow is less than or equal to 0.013 mg/L. The protective concentration of ammonia, as nitrogen [ammonia] (at 25^{o}C and Ph 8), for silvery minnow is less than or equal to 3.09 mg/L for larvae and less than or equal to 9.3 mg/L for post-larvae." IN RPA EE, the Service establishes that, "action agencies, in coordination with parties to the consultation, shall provide funding for a comprehensive water quality assessment and monitoring program in the Middle Rio Grande to assess water quality impacts on the silvery minnow. This assessment and monitoring program should use available data from all sources." The Service is conducting a water quality assessment along the MRG in conjunction with silvery minnow habitat use. This study is assessing concentrations of a wide selection of both organic and inorganic constituents.

3.1.2.5 Competition and Predation

The silvery minnow evolved with natural predation and competition (BOR and COE, 2003). According to the Service (FWS, 1994), the Rio Grande chub and bluegill are native predators of silvery minnows. Exotic species introduced into the Rio Grande starting in the late 1800's may have changed the dynamics of these interactions, however. Potential non-native predators include the northern pike, walleye, white crappie, white bass, black bullhead, brown bullhead, smallmouth bass, and largemouth bass. Other exotic fish species that may prey on silvery minnow include channel catfish, flathead catfish, and red shiner.

Competition and predation interactions among native and introduced fish species and the silvery minnow in the Rio Grande are poorly known, but need to be considered to avoid unwittingly promoting the habitat of potential predators. For instance, the red shiner, which aggressively preys on the early life-stages of many species, spawns in warm, shallow, low-velocity, sandy habitats (Sublette et al., 1990; Nico and Fuller, 2000; Burkhead and Huge, 2002), conditions similar to those currently viewed as benefiting the silvery minnow. Studies to assess potential predatory relationships between red shiner and other exotic species and early life-stage silvery minnows have not been conducted.

3.1.3 Population Ecology

Ultimately, the success of habitat rehabilitation efforts must be gauged by the response of the silvery minnow population. Currently, the numbers of wild silvery minnows are unknown (FWS, 2003a), and monitoring methods used in the MRG are not appropriate for making population estimates (Platania and Dudley, 2003). Low population density, patchy distribution (i.e., tendency for schooling), small number of sampling sites, and sampling protocols that lack statistically based designs preclude making reliable population estimates. These issues notwithstanding, trends in silvery minnow population are inferred from seine netting at a number of sites between Bernalillo and Elephant Butte Reservoir (Fig. 3-1).

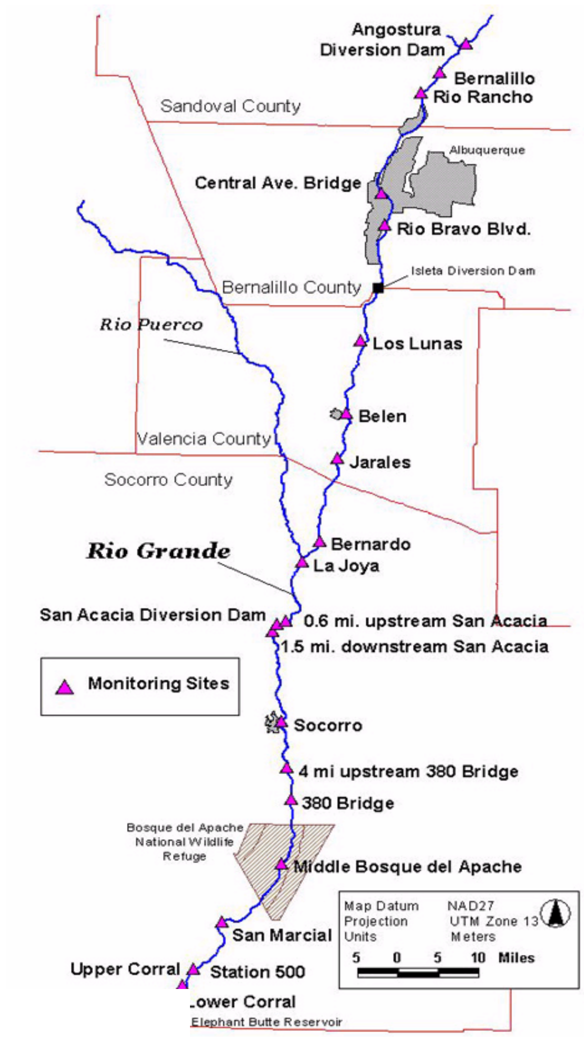

Figure 3-1: Silvery minnow monitoring site map for MRG

Monitoring data collected over the last decade suggest that, with the exception of 2002, the population was generally higher in the southern reaches than the northern reaches (Fig. 3-2). In addition, these population trend data illustrate substantial year-to-year variability, where order of magnitude changes in capture numbers are not uncommon. From the perspective of population ecology, the boom and bust population trend is common among organisms adapted to ephemeral environments with high degrees of resource variability. Such species tend to be small, relatively short-lived, and produce large numbers of eggs (or seeds in plants); mortality rates tend to be independent of population size and generally related to extremes of environmental variability (Ricklefs, 1973). The potential for high reproduction rates allow these populations to grow rapidly under favorable environmental conditions. Dispersal mechanisms work to increase the likelihood that offspring will find suitable habitats in ephemeral environments.

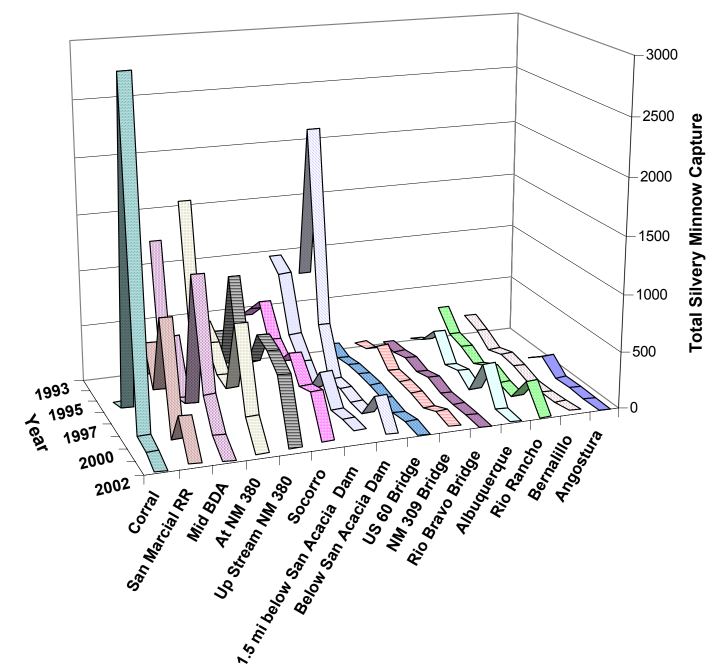

Figure 3-2: Silvery minnow capture data between Corral and Angostura
(Data from the BOR silvery minnow-monitoring database)

The 2003 BA suggests that one factor that currently places the silvery minnow at greatest risk is low population densities (BOR and COE, 2003). Low population densities are of concern because it reduces the chance of finding suitable mates. Continued decreases in population to levels below the minimum viable population threshold impede recovery efforts, even with increased habitat availability. The importance of the minimum threshold level concept is controversial for schooling fish like the silvery minnow, however, because adequate spawning partners can always be found within their school. The potential for large swings in population complicates the interpretation of the population dynamics relative to establishing the minimum viable population thresholds and evaluating the effects of restoration practices. Additional assessment is needed to determine to define the viable population size for this species (BOR and COE, 2003).

The 2003 BA provides three hypothesized patterns of population response deduced from the available silvery minnow data (BOR and COE, 2003). At moderate densities (i.e., catch per unit effort [CPUE] of 51-150 silvery minnows per 100 m^2), the populations appear to be regulated by increasing productivity of young with decreasing population densities. Under environmentally benign conditions, these moderately dense populations generally maintain relatively stable numbers or grow. Low-density populations (35-50 silvery minnows per 100 m^2) appear unable to attain/regain a higher density class, apparently because of reduced reproductive capacities due to the declining densities. At-risk populations (< 35 silvery minnows per 100 m^2) have subtle fluctuations in stock numbers, with declining trends in density, leading to eventual collapse and extirpation of the stock. Additionally, all three population categories may suffer large population losses caused by extreme environmental conditions (e.g., channel drying) that can result in localized collapse or extirpation in a relatively short time. A fourth population category (>150 silvery minnow) lacks contemporary time-series data on which to characterize population responses (BOR and COE, 2003).

3.1.3.1 Life Expectancy

In the wild, Silvery minnows rarely survive many months beyond their first reproductive period, near age-1 (FWS, 2003b). Typically, age-2 or older fish comprise less than 10 percent of the spawning population of silvery minnows (FWS, 2003b). In captivity, a majority of the silvery minnow stock live beyond a year, with some surviving up to 5 years. However, anecdotal information indicates that widely divergent mortality rates captive and wild fish are not uncommon. The dominant causes of mortality of silvery minnows in the wild have not been documented, but may be related to episodic channel drying, inadequate food supplies, predation, and disease.

3.1.3.2 Growth

Development and growth rates for wild silvery minnows are not well defined, but appear to vary in response to water temperatures and other environmental variables. Platania (1995) indicates that eggs, 3 mm in size, hatch within 1 to 3 days; the protolarvae grow to about 6 mm over the next 1 to 2 days. The silvery minnow develop through meso- and metalarvae stages, reaching the post-larval juvenile stage (about 15 mm in length) within about 50 days. Subsequently, juveniles can grow to as large as 70 mm by 240 days post hatch, with mature, spawning minnows ranging from about 35 to over 80 mm at the end of the first year (Dudley and Platania, 1997). Sublette et al. (1990) report the maximum length for silvery minnows of about 90 mm. Most of this growth likely occurs between July and October, because lower winter temperatures limit food production and feeding activity by fish.

3.1.3.3 Reproductive Biology

For fish inhabiting rivers susceptible to periods of desiccation, understanding the timing and duration of spawning, egg dispersal, and upstream movement is important for developing appropriate restoration strategies. Silvery minnow spawning can occur from April through at least June, with the peak egg production occurring in mid to late May (Platania and Dudley, 2002a, 2002b; Fig. 3-3), coinciding with snowmelt runoff. Peak egg production occurs over about 3 days with sporadic or low-level spawns occurring over the next 4 to 6 weeks (Platania and Dudley, 2002a, 2002b). The conditions that trigger spawning are not completely understood, but peak spawns are generally correlated with increases in flow volume and turbidity levels in the spring (Platania and Dudley, 2002a, 2002b). The minimum volume of flow needed to initiate spawning is unknown, but significant spawns have been observed with flows as low as 500 to 600 cfs, and minor spawns have been observed with no apparent increase in flow (Platania and Dudley, 2002a, 2002b). Following the peak spawn, increased flows do not consistently trigger significant egg production. Temperature, degree-days, sediment (turbid water), and photoperiod have been suggested as other possible triggering mechanisms, although their role in initiating spawning has not been experimentally demonstrated.

Dudley and Platania (1999b) report that minnow eggs are slightly negatively buoyant, with a specific gravity of 1.0059±0.0001 at 20º C in distilled water and a critical settling velocity of about 9.3 mm/s. The relative buoyancy of the eggs would increase as the suspended sediment concentration increases and temperature decreases. Because the eggs are only very slightly negatively buoyant, only minor currents, such as those generated by winds, would be required to keep the eggs in suspension.

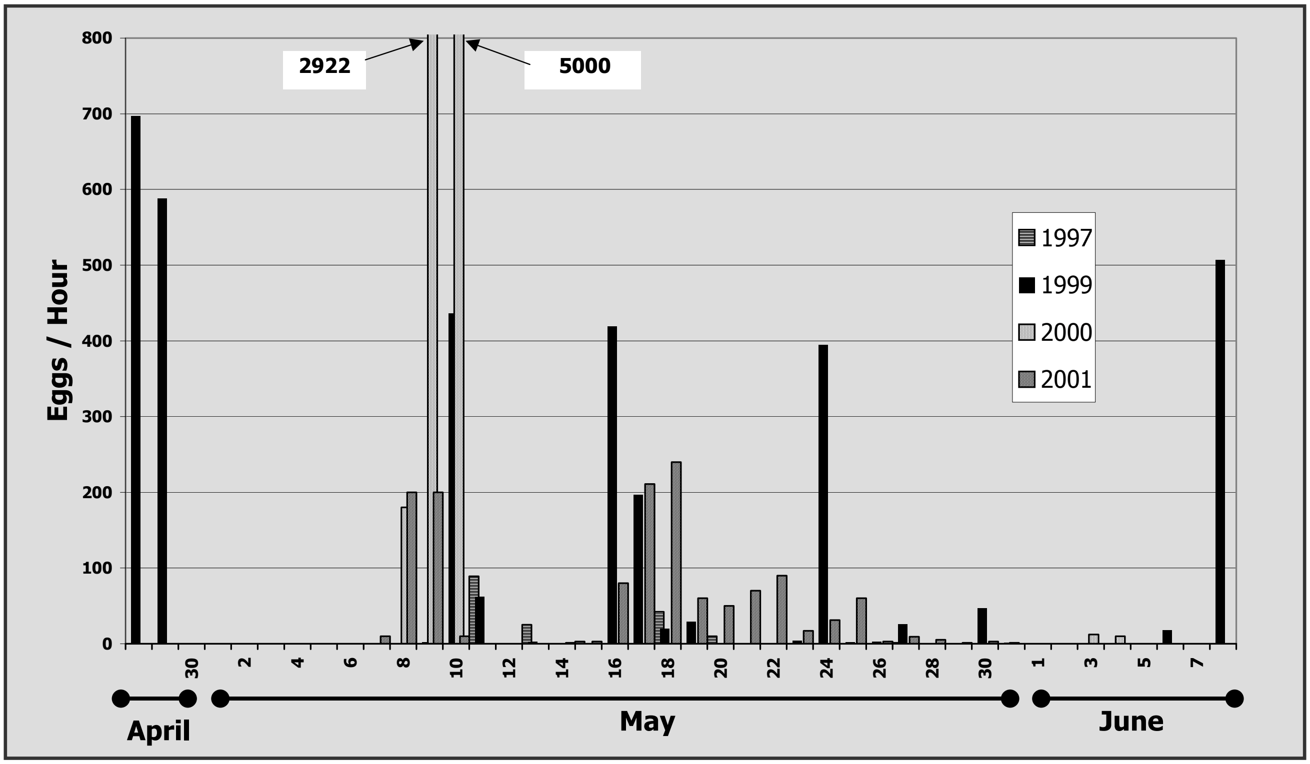

Figure 3-3: Analysis of silvery minnow spawning peaks downstream from San Marcial
(BOR and COE, 2003)

3.1.3.4 Egg Dispersal

Pelagic broadcast spawning is hypothesized to enhance the reproductive success of some fish species by reducing egg burial and suffocation in rivers with shifting sand beds (Araujo-Lima and Oliveira, 1999). The BA suggests that this reproductive strategy was well adapted to the pre-development aquatic habitats of the MRG (BOR and COE 2003). The pelagic mode of spawning evolved under channel conditions with greater diversity in flow velocities and hydrologic retention during flooding, which helped minimize the downstream displacement of eggs and developing larvae. These effects would have been particularly important in the upstream reaches of the MRG that have more reliable water supplies.

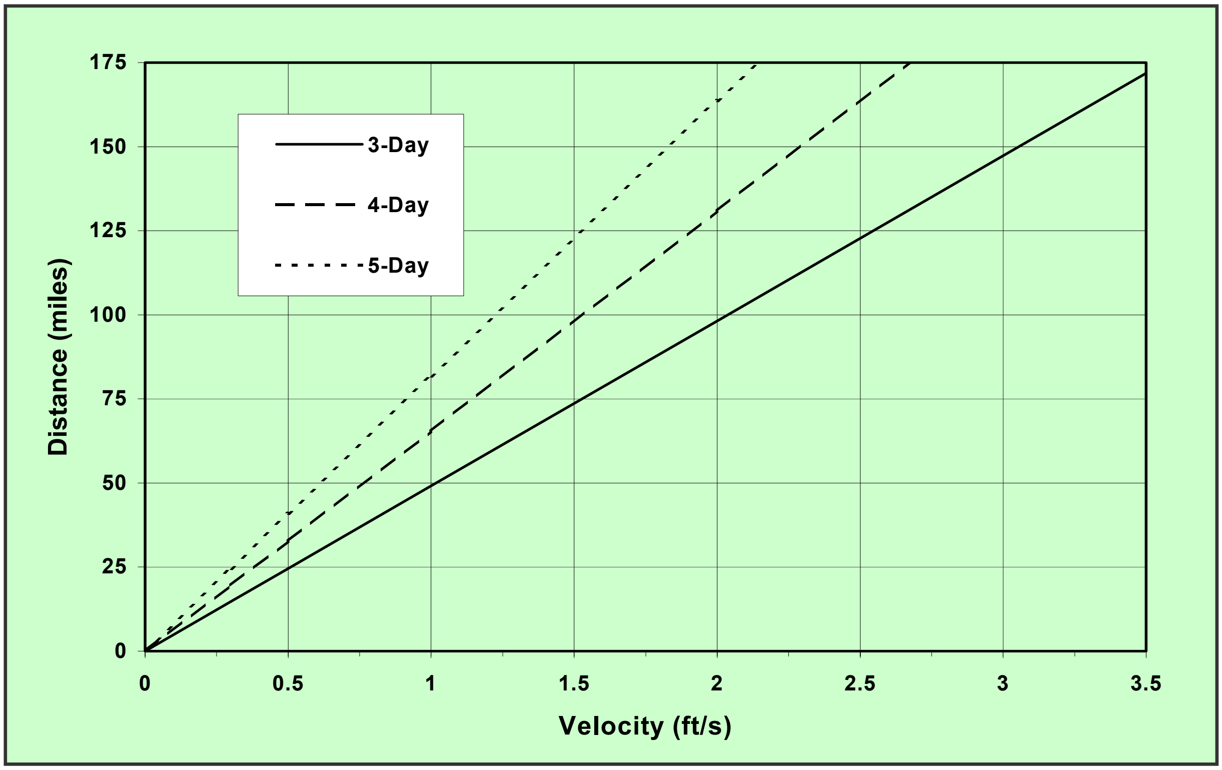

Figure 3-4: Relationship of water velocity to potential distance of egg transport

Under current conditions, however, rather than providing silvery minnows with a survival advantage, pelagic broadcast spawning now appears to be detrimental. River maintenance and channel straightening activities in the MRG have reduced the amount of low velocity environments and the potential for retention of eggs and larvae in the upper reaches. Platania and Altenbach (1998) estimated that eggs and larvae entrained mid channel could potentially be transported 100 to 200 miles downstream in the 3 to 5 days required for the minnow to develop swimming abilities after fertilization of the eggs. The actual transport distances are unknown, but the current distribution of fish suggests that the existing conditions in the river have produced net downstream displacement of the population. The distance of downstream displacement depends on current velocities, and potential retention of eggs and larvae in slack water areas, including backwater eddies, side channels, or submerged edges of the floodplain.

The implications of flow velocity with regard to the transport of eggs and larvae are illustrated in Figure 3-4, which shows the transport distances under a range of average water velocities for the 3- to 5- day period. Flow velocities can vary with discharge, channel configuration, and location within the channel, ranging from near 0 to more than 8 ft/s in association with a typical spring discharge peak (≥4,000 cfs, Miller and Mussetter 2003). Recognizing that the flow velocity environment is complex, it is clear that very low mean velocities are needed to retain the eggs and nonmotile larvae within a relatively restricted range. Thus, habitat features that promote slower flow conditions, especially those producing net-zero flow velocities, are important for egg and larval retention. The dispersal and fate of eggs is important to silvery minnow survival, especially since flows tend to carry the drifting life stages into irrigation diversions and ultimately into Elephant Butte Reservoir.

3.1.3.5 Upstream Swimming Ability

The capability of silvery minnows to swim upstream is not well documented. Laboratory studies reveal that adult silvery minnows have strong swimming abilities and high endurance; on par or even exceeding those found for salmon and rainbow trout, on a body length basis (Bestgen et al., 2003). The spawning drive is a common stimulus for upstream movement of many fish and it may underlie the swimming abilities of silvery minnow (K. Bestgen, Reclamation presentation). Alternatively, the strong swimming abilities displayed by the silvery minnow could function to hold them "in place" under higher velocity conditions (rheostasis, i.e., the tendency of fish to swim into the current) and upstream movement may be only coincidental. Considering that swimming abilities are probably not fully developed until late summer and observations that silvery minnow "hold-up" in lower velocity waters in the winter (Dudley and Platania, 1997), we may deduce that most upstream movement is likely to occur either in mid to late fall, before the water markedly cools, or in late winter to early spring prior to peak runoff. The observation by Platania et al. (2003), who found upstream movement between January and April by at least some captive reared silvery minnow marked and released into the MRG, lends support to the latter of these two options. Whatever the relationship, because most silvery minnows in the wild only live about one year, significant annual upstream movement of young-of-year would be required to repopulate the natal areas wherever the majority of the population is displaced downstream.

In early January of 2002, approximately 12,000 captive reared and marked silvery minnows were distributed and released between two locations along the San Acacia Reach (Platania et al. 2003). Of these fish, 66 were subsequently recaptured. For those released at the Lower Corral site, all but 2 of the 35 recaptured fish were found downstream, the remaining two were captured near their release site. Platania et al. (2003) explained that river releases of captive-reared fish typically result in high percentages of the downstream displacement; presumably due to the lack of conditioning and experience in dealing with natural flow conditions. Of the 31 recaptured silvery minnows that had been released at the Socorro site, 17 were captured 0.6 to nearly 11 miles (average 3.6 miles) downstream of the release point and 14 were captured upstream. Twelve of these fish traveled 0.6 to 5 miles upstream (average 2.0 miles), but one traveled over 16 miles upstream and one 18 miles upstream by collection days 105 and 133, respectively (all travel distances recomputed from Table 1 of Platania et al., 2003).

These results indicate that at least some silvery minnows have the ability and predisposition to swim considerable distances upstream. Whether wild silvery minnows have similar or greater swimming abilities and predispositions to move upstream and the proportion of the wild fish possessing these traits remain questions for future work. Specifically, future field studies are needed to understand the timing, capabilities, and predisposition of wild silvery minnows to swim upstream. These attributes have important implications regarding the impacts of habitat fragmentation associated with irrigation diversion structures.

3.1.3.6 Food Habits

The specific food habits of the silvery minnow are poorly defined. Qualitatively, the silvery minnow diet is considered comparable to closely related species that primarily feed on diatoms, algae, larval insect skins, and plant material contained in the ooze of bottom sediments (Sublette et al., 1990). The contents of the few silvery minnow stomachs examined tend to support this contention. Larval and adult silvery minnows apparently have similar diets, although algae are considered somewhat more important than other foods in the early life stages.

The historic channel of the Rio Grande probably contained appreciable amounts of large woody debris. Maintenance activities have removed woody debris from the channel. Relatively stable substrates, such as woody debris, provide locations for attachment and growth of algae and points of accumulation for drifting leaf litter, two of the primary food materials for silvery minnows (e.g., Sublette et al., 1990).

Organic debris contributions to the river from the riparian community have probably decreased in association with channelization, incision, and reduced lateral continuity. The adequacy of food resources for the silvery minnow in the MRG has not been documented.

3.1.4 Habitat Relationships

Aquatic habitat features are commonly divided into three categories based on size. *Microhabitat* components are features less than a few meters or a few square meters in size, *mesohabitat* components range from a few meters to about 100 meters in size (commonly, about equal in length to the channel width), and *macrohabitat* features include larger scale habitat components (Bovee et al., 1998). Microhabitats in river environments include features that have relatively homogeneous conditions of depth, velocity, substrate and cover, including undercut banks, leaf packs, and eddy areas between sand ripples, along bar faces, and behind cobbles and snags. Mesohabitat features include bars, pools, riffles, runs, embayments, oxbows, scour holes, backwaters, tributary mouths, bends, braids, and islands. Macrohabitat features include characteristics of reaches, sub-basins, and entire drainage basins like water quality, water temperature, and flow regime.

In general, fish populations (abundance) are rarely regulated by the mesohabitat attributes of local flow velocities, water depth, and substrate composition, unless they become severely limiting (e.g., abnormal increases in average flows, dewatering, or excess loads of fine sediment). For instance, large year-to-year variations in fish abundances are often observed in waters where apparent habitat availability remains relatively constant, and in cases where fish populate presumably marginal habitats in high abundance, while being apparently absent at other times (BOR and COE, 2003). In general, short-term changes in local flow velocities, water depth, and substrate more often affect the distribution of fish species rather than their abundance (BOR and COE, 2003). This presumably also applies to the silvery minnow in the MRG. Thus, it would appear that loss of native species in the MRG is due primarily to macrohabitat alterations, including factors associated with channelization, flow regulation, and longitudinal disruptions that produced an overall simplification and fragmentation of the habitat, and to microhabitat alterations that limit retention during the downstream transport of eggs and larvae. Channelization and subsequent maintenance activities along the MRG have reduced the complexity of the channel and the amount of net-zero velocity habitat. Additionally, these have removed the diversity of flow-impeding channel structures, such as woody debris, islands, and high-flow ephemeral side channels, likely resulting in greater proportions of silvery minnow eggs and larvae being transported increasingly greater distances downstream.

3.1.4.1 Habitat use by juvenile and adult silvery minnows

Our understanding of habitat use by the silvery minnow comes primarily from observations made at collection sites where the fish were captured under contemporary field conditions and from comparisons to habitats of related species in other river systems. These habitat use patterns, commonly interpreted to be habitat preference requirements for the silvery minnow, differ somewhat over time and reflect the conditions that prevailed during the individual collection efforts. Koster (1957) described the habitat of the silvery minnow as, “pools and backwaters of the main rivers and creeks” where they schooled and fed “largely on bottom mud and algae.” Sublette et al. (1990) reported that while the silvery minnow tolerates “a wide variety of habitats, it prefers large streams with slow to moderate current over a mud, sand or gravel bottom.” Bestgen and Platania (1991) observed that most silvery minnows “were captured in low velocity habitats that had sand substrate.” Platania (1991) reported that “large collections” of silvery minnows occurred at sites with “a shifting sand-silt substrate.” Watts et al. (2002) reported that silvery minnow more commonly used shoreline habitats with debris than open-water habitats lacking debris. Because these studies were all conducted in an altered river system it is unknown whether they represent optimum habitat requirements for the silvery minnow.

Overall, habitats used by juvenile and adult silvery minnows appear very similar. Young silvery minnows appear to inhabit primarily low velocity habitats (Dudley and Platania, 1997). These are areas where algae and other organisms could accumulate in eddies with drifting organic material to provide food for the young silvery minnows. As their mobility increases with age, silvery minnows are able to venture into higher velocity waters. Adults have commonly been captured in habitats having relatively low flow velocities (10 cm/s or less), and are consequently associated with shallow water (<16 inches; 40 cm) and sand-silt substrates (Dudley and Platania, 1997). In turn, the FWS (2003a) defines silvery minnow habitat as "shallow waters with a sandy and silty substrate that is generally associated with a meandering river that includes sidebars, oxbows, and backwaters.... Adult silvery minnow occur in shallow braided runs over sand substrate, but rarely in habitat with substrate of gravel or cobble."

The field studies of Dudley and Platania (1997) revealed that silvery minnows use deeper waters and coarser substrates when available. Observations made at the naturalized refugium at the Albuquerque Biological Park suggest that the silvery minnows commonly concentrate in the deep pool (K. Ferjancic, FishPro, personal communication). Laboratory studies of the swimming abilities of silvery minnows indicate that they use low velocity zones behind large cobbles to escape stronger surrounding currents (Bestgen et al., 2003). These authors reported that "cobble appears to have a number of benefits. First, observations showed that silvery minnow were not averse to swimming over it. Second, silvery minnow used the boundary layer and low velocity zones behind cobble to rest or make upstream progress. Third, cobble may provide a more natural array of cover that resting fish could seek refuge in" (Bestgen et al., 2003).

The Service has proposed four primary constituent elements of critical habitat for the silvery minnow (FWS, 2003a,b):

1. A hydrologic regime that provides sufficient flowing water with low to moderate currents capable of forming and maintaining a diversity of aquatic habitats, including backwaters, shallow side channels, pools, eddies, and runs.
2. The presence of eddies created by debris piles, pools, or backwater, or other refuge habitat with unimpounded stretches of flowing water of sufficient length that provide a variation of habitats with a wide range of depth and velocities;
3. Substrate of predominately sand and silt; and
4. Water of sufficient quality to maintain natural daily and seasonally variable water temperatures in the approximate range of greater than 1°C (35°F) and less than 30°C (85°F) and reduce degraded conditions (e.g., decreased dissolved oxygen, increased pH).

3.1.4.2 Velocity and Depth

Dudley and Platania (1997) studied the relationship among silvery minnow occurrence and water depth, flow velocity, and substrate. They reported that habitats where silvery minnows were collected differed somewhat with fish size. The majority of silvery minnows captured across all size classes were taken from waters with velocities of less than 10 cm/s. Immature fish (up to 1.1 inch [30 mm] in length) were most commonly found in waters that were less than about 12-inches (30-cm) deep, whereas the majority of silvery minnows up to about 3-inches (70-mm) in length were collected from waters that were less than about 16-inches (40-cm) deep. Silvery minnows greater than 3-inches (70-mm) in length were common in waters that were up to about 24-inches (60-cm) deep. Younger fish were often found in backwater pools with silt and sand bottoms, whereas mature fish (1.6 to 3 inches, 40 to 70 mm) were more prevalent in deeper waters (16 to 20 inches, 40 to 50 cm), particularly in areas with debris and greater proportions of sand and gravel. These differences may relate to the development of swimming strength with size and the comparative food availability among habitats. Overall, during these studies, most silvery minnows were collected at locations with depths ranging from 4 to 20 inches (10 to 50 cm) and with flow velocities of less than 10 cm/s.

Silvery minnow captures are generally associated with deeper waters during the winter compared to the summer; maximum numbers of silvery minnows were captured in the 12 to 16 inch (31 to 40 cm) depth interval in the winter compared to the 4 to 8 inch (11to 20 cm) depth interval in the summer. During the winter, more than 50 percent of the captured silvery minnows were in waters with no measurable flow; whereas during the summer, over 50 percent of the silvery minnows were captured in waters with flows of 1 to 10 cm/s. Common winter habitat was reported to be slow flowing water (0 to 4 in/s, 0 to 10 cm/s) with moderate depth (4 to 20 inches, 11 to 50 cm) associated with debris piles along shorelines (Dudley and Platania, 1996 and 1997).

Results from a recent habitat modeling study indicate that slow-flow, shallow-water habitat, commonly characterized as the preferred habitat for juvenile and adult silvery minnow (FWS, 2003b), exists in abundance in some reaches in the MRG even during flows of 4,000 cfs (Fig. 3-5, from Miller and Mussetter, 2003).

3.1.4.3 Substrate

The assumption that silvery minnows only utilize fine-textured substrates (sands and silts) is inconsistent with their historic occurrence in the Rio Chama and upper reaches of the Rio Grande. The BOR and COE (2003) described the reach of the Rio Grande between Otowi and Bernalillo as "a critical faunal transition zone," where the resident population of silvery minnows "factored prominently in the probability of colonization of empty downstream habitat patches." Early studies (pre-Cochiti Reservoir) indicate that gravel and cobbles were prevalent in the upper reaches of the Rio Grande, with gravel becoming less abundant below Albuquerque (Rittenhouse, 1944; Culbertson and Dawdy, 1964; Nordin and Beverage, 1964).

The preference of silvery minnows for habitats with fine-textured bed materials has not been demonstrated in modern studies. Dudley and Platania (1997) indicated that silvery minnows were collected over various substrates in proportion to the relative availability of the substrate at a particular site. Thus, silvery minnows were associated with gravel and cobble substrates where these conditions existed (e.g., Rio Rancho site) suggesting that the species does not exhibit a particular substrate preference. The occurrence of finer-textured substrates in the lower reaches is related to the downstream fining processes, whereby sediment size decreases with distance from the higher-gradient tributary watersheds. Because finer-textured bed materials are more prevalent in the reaches currently occupied by the silvery minnow, the increased capture frequencies over silt and sand substrates may be explained for purely probabilistic reasons.

The historical occurrence of rock and gravel spawning species, such as the shovelnose sturgeon, blue sucker, and gray redhorse, and the presence of long-nose gar in the MRG (Sublette et al., 1990) suggests that coarse substrates were common substrates in the system, meeting these species specialized reproductive requirements. The extent to which the silvery minnow may have used these areas historically cannot now be estimated, but there is little basis to suspect that the structure of the benthic substrate directly affects population numbers of the pelagic silvery minnow. Indeed, silvery minnows "are neither habitat specialists, benthic, nor sedentary but rather they are inhabitants of the water column" (Platania et al., 2002:26).

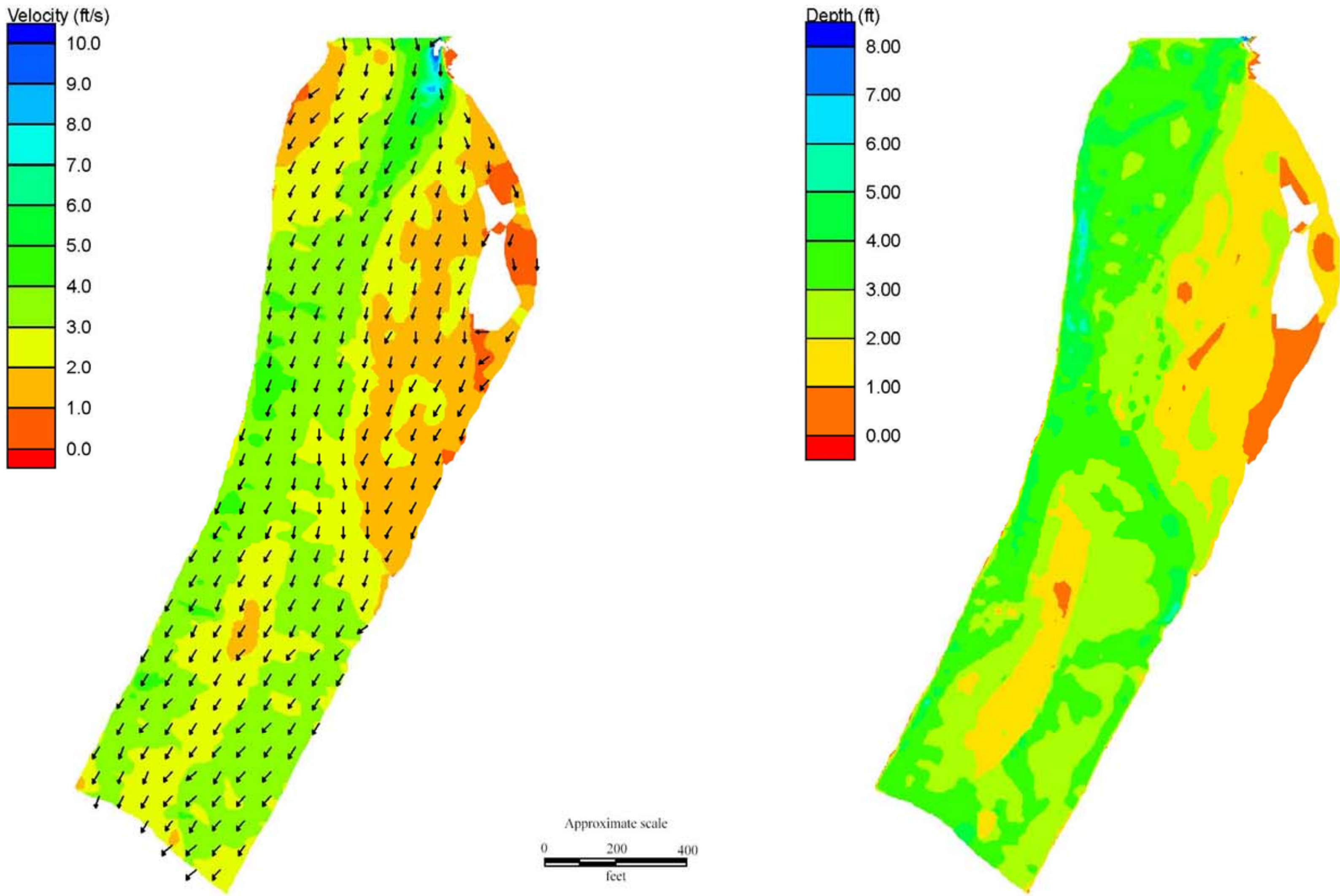

Figure 3-5: Flow-velocity and depth profiles at the Bernardo pilot modeling site.
(Miller and Mussetter, 2003)

Gravel-cobble substrates are inherently more productive habitats for algae, invertebrates, and fish than sand-silt beds. Silt and sand are marginal substrates for periphytic algal growth and benthic invertebrate colonization. Stable substrates, including gravel, cobble, and woody debris, for benthic algae and invertebrate colonization are generally lacking within the current range of the silvery minnow. Instead, most such growths are transitory, associated with relatively scarce backwaters, or eddies downstream of metastable sand bars, dunes, and ripples. Feeding on accumulations in such areas, resulting in perhaps incidental and conspicuous consumption of sand with the algae, detritus, and other target food particles, and may account for the reports of abundant sand among the gut contents of silvery minnows in contemporary collections (e.g., Sublette et al., 1990). Sand and silt are unlikely to be essential diet items for silvery minnow.

In aggregate, this information strongly suggests that silvery minnows are adapted to a broad range of bed conditions. Thus, sand and silt substrates may represent an accessory characteristic of silvery minnow habitat rather than a differentiating criterion. The interdependence of flow velocity, water depth, and substrate characteristics complicates the interpretation of the substrate requirements for the silvery minnow (BOR and COE, 2003). The size of the bed material is generally dependent on flow velocity, rather than being an independent variable (Leopold et al., 1992, Gordon et al., 1992). However, this relationship may be affected by source area considerations, whereby coarse bed materials can occur in low velocity environments and finer materials can occur under higher velocity flow regimes in response to local source conditions.

3.1.5 Summary of Habitat Priorities for the Silvery Minnow

From a broad-scale perspective, based on information compiled in the previous subsections, the conservation and recovery of wild populations of silvery minnow in the Cochiti Reservoir to Elephant Butte Reservoir reach of the MRG will require addressing, at minimum, 6 limiting factors currently affecting this species. The suite of actions necessary for habitat restoration to meet the goal of egg and larvae retention and young-of–year rearing habitat are listed below and discussed in greater detail in Section 5.1.

- Sustained flows in key reaches to promote sufficient populations of wild silvery minnows
- Spring flow peak in mid- to late-May to stimulate spawning
- Establishment of channel conditions that retard downstream displacement of eggs and larvae
- Establishment of a sustainable population of silvery minnows in the Angostura reach
- Establishment of suitable feeding and cover habitat for juveniles and adults
- Remediate longitudinal discontinuity associated with irrigation diversion structures

3.2 Southwestern Willow Flycatcher

The endangered southwestern willow flycatcher is a subspecies of flycatcher with a breeding range throughout much of the southwestern United States and a winter range that includes Mexico and Central America. Habitat restoration efforts along the MRG would tend to benefit the breeding season and migration periods. The successful conservation and recovery of the flycatcher will depend on addressing impacts to this species within the MRG and at locations away from the influence of the Program. The following sections present information regarding the status and distribution of the flycatcher and the contemporary environmental stresses that affect it, as well as population ecology and habitat relationships. Finally, priority needs for restoration along the MRG are described.

3.2.1 Status and Distribution

The flycatcher is a small, migrating Neotropical, passerine (perching) bird about 6 inches (15 cm) long. It is one of 11 *Empidonax* species that breed in North America (Sogge, 2000). This subspecies is distinguished based on morphology, song type, habitat use, structure and placement of nests, ecological separation, and genetic distinctness.

The flycatcher was listed by the Service as endangered due to "extensive loss of habitat, brood parasitism, and lack of adequate protective regulation" (FWS, 1995). The states of New Mexico, Colorado, California, Texas, and Utah list the flycatcher as endangered, while the State of Arizona includes it on its draft list of Wildlife of Special Concern. The State of Nevada considers the species to be Critically Impaired.

A migratory species, the flycatcher winters in Mexico and Central America and travels to the southwestern United States to breed (Finch et al., 2000). The breeding range of the flycatcher is centered in New Mexico, Arizona, and southern California, although it extends into the fringes of the adjoining states (Nevada, Utah, Colorado, and Texas) and northern Mexico. The Rio Grande valley is generally considered the eastern extent of flycatcher breeding, although a few individuals nest along the Canadian River and, perhaps, the Pecos River in New Mexico.

The modern extent of the flycatcher habitat has not markedly diminished from its historical range, but the quantity and quality of breeding habitat and its population numbers have declined. The Rio Grande from the headwaters in Colorado to the Pecos River confluence supports about 128 territories, more than 10 percent of the range-wide total for identified flycatcher territories (FWS, 2002). Thus, the Rio Grande ecosystem is important for maintaining the viability of the population.

The FWS (2002) recovery goal, which would allow downlisting the flycatcher from endangered to threatened, is the establishment and maintenance of 1,950 territories in 32 Recovery Units representing the historic range of the flycatcher. The recovery goal includes 100 territories for the Middle Rio Grande Recovery and Management Unit (Otowi Gage to Elephant Butte Reservoir Dam) and 75 territories in the Upper Rio Grande Recovery and Management Unit (the Colorado-New Mexico stateline to Otowi). These numbers represent minimum recovery goals and must be maintained over a 5-year period. Two additional criteria that are required include:

- Minimize the major stressors to the flycatcher and its habitat (including but not limited to floodplain and watershed management, groundwater and surface water management, and livestock management

- Ensure that natural ecological processes and/or active human manipulation needed to develop and maintain suitable habitat prevail in areas critical to achieving numeric stability within each population center

Overall costs for recovery are estimated to be in excess of $130 million for 20 years across the entire range; costs for the MRG recovery units were not specifically identified. Implementation of the Recovery Plan is projected to result in downlisting of the flycatcher to threatened within 10 years and delisting within 20 years (FWS, 2002).

3.2.2 Contemporary Environmental Stresses

Over the flycatcher's entire breeding range, habitat loss has been attributed to impoundments and flood control, water diversion and channelization, wildfires, urban and agricultural development, livestock grazing, invasive exotic plant communities, and recreational uses (FWS, 1995; Marshall and Stoleson, 2000). Pesticides and agricultural chemicals in irrigation return waters and sediment are also speculated to adversely affect the flycatcher, but direct evidence for this impact is lacking. Poorly managed grazing can reduce the amount and quality of breeding habitat especially in small riparian zones. While impacts from grazing within the MRG bosque are a concern, preliminary data indicate that this may not be a pervasive threat to the flycatcher; limited impacts may occur in localized areas (Ahlers, 1999).

Fire can result in direct destruction of breeding sites and shifts in plant community structure (Paxton et al., 1996; FWS, 2002). On average, about 850 acres of bosque burn per year, based on a review of fire records for the Rio Grande Valley for the 1985 to 1995 period (Stuever, 1997). Conversion of undeveloped land to agricultural or urban uses can increase disturbance stress on flycatchers (FWS, 2002 and 2003).

Common nest predators of the flycatcher that prey directly on flycatcher eggs and young include great-tailed grackles (*Quiscalus mexicanus*), magpies (*Pica pica*), and common ravens (*Corvus corax*). Brown-headed cowbirds (*Molothrus ater*) are brood parasites that destroy flycatcher eggs and then lay their eggs in the host nest to be raised by flycatchers (FWS, 2002, Ahlers and Sechrist, 2002). The grackle, raven, and cowbird are native species whose population may be favored by agricultural and urban developments.

3.2.3 Population Ecology

Throughout the MRG, 6 flycatcher population centers are recognized (Fig. 3-6). The number of territories, pairs, nest attempts, and successful nests in these areas has varied with time. To provide an indication of recent observations regarding status and trends, the Recovery Plan estimated that 51 breeding territories occurred within the Middle Rio Grande Recovery and Management Unit and 37 territories occurred within the Upper Rio Grande Recovery and Management Unit (FWS, 2002). In the 2002 survey by Reclamation, 84 territories were documented along the MRG reach, including 1 in the Belen reach, 13 in the Sevilleta/La Joya reach, 4 in the San Acacia reach, 3 in the Bosque del Apache reach, and 63 in the San Marcial reach (Moore and Ahlers, 2003). Of these, 16 nesting territories (20 percent) occurred within the Elephant Butte Reservoir delta. If territories likely to occur within the Isleta Pueblo (14 documented in 2000) and within unsurveyed areas between the San Marcial Railroad Bridge and Bosque del Apache are added, it is likely that over 100 active nests exist along the MRG between the Belen reach and Elephant Butte Reservoir (Moore and Ahlers, 2003).

Continued monitoring will be required to assess the stability of these nesting territories and whether the minimum goal for territories within the Middle Rio Grande Recovery and Management Unit is being maintained at adequate levels (FWS, 2002)

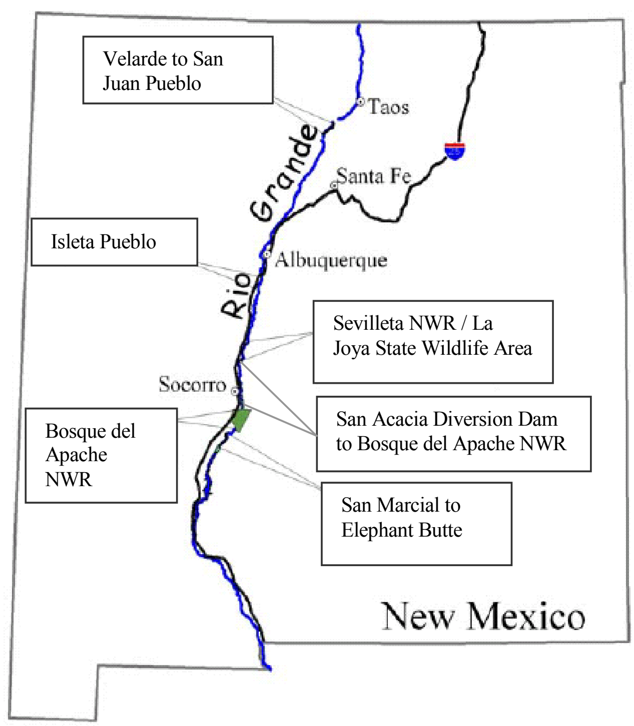

Figure 3-6: Location of flycatcher population sites
(BOR and COE, 2003)

Activities specified in the Recovery Plan to aid flycatcher recovery in the MRG include increasing the availability of water in active channels or in near channel areas and promoting the conservation and enhancement of the large local flycatcher populations, particularly in the San Marcial area (FWS, 2002). Large populations, such as those occurring at San Marcial, are seen as key elements in the recovery of the flycatcher. At this time, these populations are expected to serve as source populations that may promote future recruitment. The fate of the flycatcher population in the Elephant Butte reservoir delta just south of the Program Area is uncertain.

3.2.1.1 Life History

From wintering areas in Mexico and Central America, flycatchers begin to arrive at New Mexico breeding sites in early May. Males usually arrive a week or so ahead of females and yearlings and begin to establish territories. Flycatchers tend to return to the same general breeding area year after year, but not necessarily to the same nesting site or territory. In some instances, individuals migrate to new breeding areas in entirely different watersheds (FWS, 2002). The adults and juveniles begin their southern migration in July and August, 3 to 4 weeks after nesting. The flycatcher life span is generally 1 to 3 years, with some individuals living 4 to 7 years (Langridge and Sogge, 1997; Paxton et al., 1997; Netter et al., 1998).

3.2.1.2 Reproductive Ecology

After arriving at the breeding areas, male flycatchers establish territories that are typically 0.2 to 0.5 ha (0.5 to 1.2 acre) in size, although they can be larger depending on habitat quality and population density. Flycatchers are rarely found in patches that are narrower than about 10 m (33 feet; Sogge and Tibbitts, 1994: Sogge and Marshall, 2000). The criteria that females use to select a mate are unknown, but may be related to habitat or mate quality.

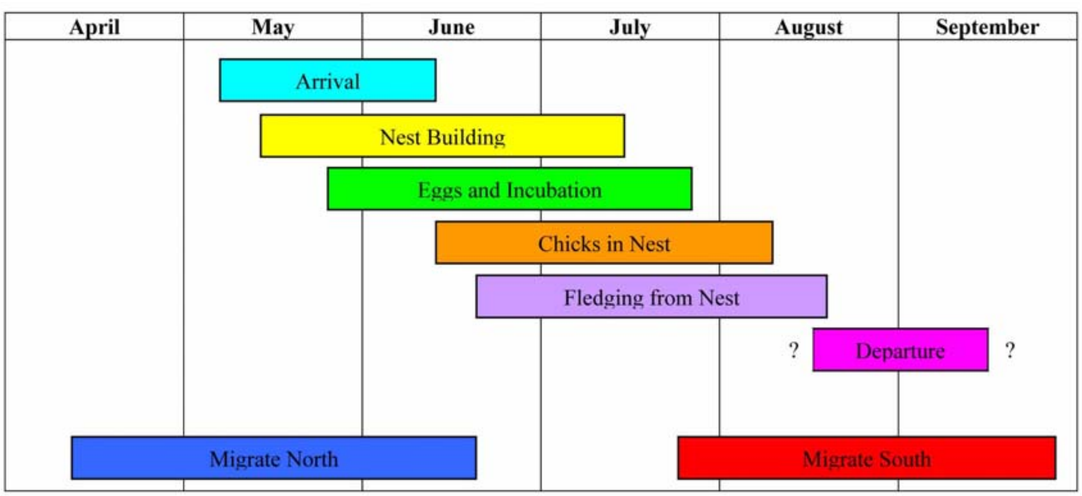

Figure 3-7: General nesting chronology for southwestern willow flycatchers
(Adapted from BOR and COE, 2003; Sogge, 2000; and FWS, 2002)

Range-wide, flycatchers build nests and lay eggs in late May and early June, with young being fledged by early July; however, these characteristics are locally affected by altitude, latitude, and renesting attempts (Fig. 3-7). Second broods or nesting attempts can occur into August. Females construct their nests of shredded bark, cattail tufts, grass, and feathers over a 4- to 7-day period. They generally lay one egg per day until there are 3 to 4 eggs in the nest (Gorski, 1969). If multiple breeding attempts are made in one season (i.e., renesting in response to failure of the first attempt due to parasitism or predation), the clutch size is typically smaller (Holcomb, 1974; McCabe, 1991; Whitfield and Strong, 1995)

The females, or rarely males, incubate the eggs for about 2 weeks. The hatchlings require about 2 weeks to mature and fledge, after which the young remain in their parents' territory for about another 2 weeks. The male and female continue to feed them during this time. Little is known about fledgling activities after this period (FWS, 2002).

3.2.1.3 Food Habits

Our understanding of the food habits and prey base of the flycatcher is still evolving (Drost et al., 2001; DeLay et al., 2002). Most flycatcher species catch insects on the wing and glean prey from foliage and the ground. As a group, flycatcher species take a wide range of invertebrate prey, including spiders, flying insects, and ground- and vegetation-dwelling insects (Beal, 1912; McCabe, 1991). Of note, the majority of these insects have terrestrial origins, as opposed to organisms with obligate aquatic stages. Typically, only a minor component of the flycatcher diet is composed of invertebrates with aquatic stages, such as dragonflies and damselflies (DeLay et al., 1999, 2002; Drost et al., 2001). Flycatchers occasionally consume small fruits, such as elderberries (*Sambucus canadensis*) or blackberries (*Rubus* species), although this is not considered an important food source during breeding season (McCabe, 1991).

Drost et al. (2001) suggests that flycatchers are dietary generalists and food shortages are unlikely to be encountered. In contrast, DeLay et al. (2002) indicated that flycatchers are selective and could be susceptible to stochastic or deterministic declines in the insect food base. The interpretation of selectivity was based on the comparison of dietary (feces) and sticky trap insect counts.

DeLay et al. (2002) recognized the limitations of sticky traps with regard to accounting for diurnal and nocturnal invertebrate activity relative to flycatcher feeding times, and capture efficiencies of flying versus crawling insects. However, they did not address prey susceptibility (catchability) issues in evaluating selectivity. Dietary data from study sites in New Mexico, Arizona, and California indicate that the most common invertebrates in flycatcher feces were from the orders Hymenoptera (bees and wasps), Hemiptera (leafhoppers), Coleoptera (beetles and lady bugs), and Odonata (dragonflies and damselflies) (Drost et al., 2001; DeLay et al., 2002). These insect groups tend to hover or crawl on branches, behaviors that would make them "easy prey" for flycatchers. The ease of capture may well explain the divergence between the feces and sticky trap data. Thus, flycatchers can be viewed generally as opportunistic generalists, feeding on insects that are easy to capture, rather than selective feeders.

DeLay et al. (1999) reported that the numbers of invertebrates, particularly flies, captured using sticky traps during two spring and one fall migration periods were significantly lower in the saltcedar habitats at Bosque del Apache National Wildlife Refuge. They also report "mixed evidence" that migrating flycatchers (but not necessarily southwestern willow flycatchers) responded to the observed variations in the insect numbers. Additional study is needed to better assess the feeding strategies of flycatchers and the potential for food limitations in the MRG during both breeding and migration periods.

Differences in insect communities undoubtedly exist among different vegetation types, although the implications for flycatchers are less certain. Drost et al. (2001) concluded that such differences do not mean that the food source from native-dominated communities is superior to mixed or exotic dominated communities. For example, the occurrence of saltcedar was found to enhance flycatcher prey availability in mixed stands, since saltcedar flowered for a longer period than native vegetation, attracting pollinators later into the breeding season (Drost et al., 2001). Drost et al. (2001) therefore concluded that efforts to remove saltcedar from some areas might result in the reduction of the high quality feeding habitats during the breeding season for flycatchers.

Owen and Sogge (2002) studied the fundamental physiological conditions of the flycatcher in native- and exotic-dominated stands with the intent of reducing speculation concerning food source differences based on insect population studies. They studied breeding habitats at six sites in Arizona and New Mexico and concluded that individuals feeding predominantly in saltcedar habitats appeared to have a higher protein-content diet relative to those feeding predominantly in native vegetation communities (Owen and Sogge, 2002). These authors state that invertebrate communities associated with some saltcedar-dominated and mixed native-saltcedar vegetation communities "may provide better energetic/dietary conditions than native habitat" (Owen and Sogge, 2002, page 20). Whether these results can be extrapolates and applied to the MRG or other flycatcher breeding areas requires additional investigation.

The available food base in flycatcher habitat can be positively or negatively influenced by the surrounding land uses. For example, mesquite stands (pollinators), emergent wetlands, and certain agricultural crops harbor or attract "tourist" insects that may travel to and enhance the food base in adjacent flycatcher habitat (Drost et al., 2001). Saltcedar flowers later into the season when flowers on other plant species are less abundant, which can attract pollinators and increase the potential food base for flycatchers.

Agricultural chemicals and pesticides are widely used in many regions through which flycatchers migrate and winter, thereby potentially exposing flycatchers to these substances (FWS, 2002). The presence and potential impacts of environmental contaminants at 10 flycatcher breeding sites in Arizona and one in California found only one pesticide (DDE) in egg samples, which occurred at concentrations not representing a hazard to flycatchers; of the array of metals assessed, only selenium and boron were found at concentrations above background, with one egg having a potentially toxic concentration of selenium (King et al., 2002). These results from sites near agricultural areas, the apparent lack of a significant source of metal contamination along the MRG, and the ongoing trend of increasing flycatcher populations along the MRG tend to minimize expectations that either pesticides or metals may be affecting MRG flycatcher populations. However, the relationships of contaminant concentrations to flycatchers along the MRG have not been studied. Therefore, such studies are appropriate as the combination of conditions producing potential toxicity may occur in some years at some locations.

3.2.4 Habitat Relationships

The flycatcher is considered a riparian obligate (FWS, 2002). Historically, flycatcher habitat along the Rio Grande consisted primarily of thickets of willows and seepwillow with an overstory of scattered cottonwood (Grinnell and Miller, 1944; Phillips, 1948; Unitt, 1987). Breeding habitat currently used by flycatchers along the Rio Grande consists of native and non-native plant communities. In addition to nesting in both Goodding's and coyote willows, flycatchers in the MRG will build nests in saltcedar and occasionally Russian olive (Moore Ahlers, 2003). Nesting success rates are comparable between flycatchers using saltcedar-dominated habitats and those nesting in native vegetation (Sferra et al., 2000). The largest concentration of breeding territories along the MRG occurs in mixed stands of Gooddings willow, cottonwood, and saltcedar in the delta area of Elephant Butte Reservoir (Moore and Ahlers, 2003).

Flycatchers are most commonly found in vegetation patches that are 0.6 ha (1.5 ac) or larger, with the vegetation used for nesting ranging in height from about 2 to 30 meters (6 to 100 feet) (Sogge et al., 1997). Regardless of plant species composition or height, occupied sites usually consist of dense vegetation in the patch interior, or an aggregate of dense patches interspersed with openings. This dense vegetation occurs mostly within the first 3 to 4 meters (10 to 13 feet) above the ground, and the dense patches are commonly interspersed with small openings, open water, or shorter/sparser vegetation, creating a mosaic that is not uniformly dense (FWS, 2002). Cottonwood gallery forests that are devoid of an understory and appear park-like do not provide breeding habitat for flycatchers (FWS, 2002). Also key to successful nesting is the presence of upward-prong, multi-twig structures for nest placement. As most willow species mature, the prevalence of this twig structure and the suitability of the plants for flycatcher nests declines. In contrast, the structure and the availability of suitable nest sites tend to persist in maturing saltcedar (FWS, 2002).

Moore and Ahlers (2003, page D-16) concluded from the 2002 surveys for Reclamation that the "[r]esults of this study show that a significant amount of highly suitable flycatcher habitat exists within the Middle Rio Grande, primarily within the San Marcial reach. Data gathered during the 2002 field season planned to update the current habitat model to show habitat for the entire MRG as well as update habitat formerly classified within the southern reaches. This reclassification will likely show an increase in the amount of highly suitable and moderately suitable habitat within the delta of Elephant Butte Reservoir."

3.2.4.1 Flooding and Food Production

Proximity to water appears to be correlated with flycatcher habitat; however, the mechanism underlying this relationship is not definitively known. Based on studies conducted in New Mexico, Arizona and California, including Bosque Del Apache National Wildlife Refuge, DeLay et al. (2002) concluded that the flycatcher's association with water is not food-based. As discussed earlier, except for the occasional consumption of dragonflies and damselflies, insects with aquatic-stage life cycles are rarely found in flycatcher feces, and much of the invertebrate prey base has terrestrial origins, often associated with litter layers (DeLay et al., 2002; Drost et al. 2001).

Repeated or prolonged periods of flooding should be expected to reduce the potential short-term production of food for flycatchers by drowning terrestrial invertebrates. Experimental flooding along the Rio Grande revealed a short-term reduction in the number of terrestrial invertebrates following flooding (Ellis et al., 1996). Over the longer term, the experimental flooding resulted in a shift in composition of the invertebrate community, but had little clear effect on overall total numbers or total productivity (Ellis et al., 1996). Additional investigation is required to determine the relationship of flooding to invertebrate production and subsequent flycatcher breeding success.

3.2.4.2 Habitat Relationships to Flooding

Johnson et al. (1999) hypothesized that flycatchers will not nest in the absence of flowing water and that overbank flooding might be important to encourage flycatcher nesting. These hypotheses are not well supported by more extensive observations. For example, flycatcher pairings and successful nesting commonly occurs in both flooded and unflooded territories, and unsuccessful pairings and nest failures occur in flooded territories (Moore and Ahlers 2003). Successful nesting occurs at sites where surface water may be present early in the breeding season, but only wet soil remains by late June and through the balance of the breeding season (FWS, 2002). In other instances, surface water is present or nearby in some, but not all years (e.g., during droughts or where reservoirs recede).

The flycatcher Recovery Plan (FWS, 2002, page 34) reports that "Surface water diversions and groundwater pumping for agricultural, industrial, and municipal uses are the major factors in the deterioration of southwestern willow flycatcher habitat. The principal effect of these activities is the simple reduction of water in riparian ecosystems and associated subsurface water tables." Also, the Recovery Plan states that:

> "the apparent association between southwestern willow flycatcher habitat and quiet water likely represents the relationship between the requirements of the bird for certain vegetation characteristics and patch size/shape, and the hydrological conditions that allow those conditions to develop.... By definition, the riparian vegetation that constitutes southwestern willow flycatcher breeding habitat requires substantial water. Further, hydrological events such as scouring floods, sediment deposition, periodic inundation, and groundwater recharge are important for flycatcher's riparian habitats to become established, develop, and be recycled through disturbance. It is critical to keep in mind that

> in the southwest, hydrological conditions at a [flycatcher nesting] site can vary remarkably within a season and between years. At some locations, particularly during drier years, water or saturated soils are only present early in the breeding season.... At other sites, vegetation may be immersed in standing water during a wet year, but be hundreds of meters from surface water in dry years.... Similarly, where a river channel has changed naturally..., there may be a total absence of water or visibly saturated soil for several years. In such cases, the riparian vegetation and any flycatcher breeding within it may persist for several years" (FWS, 2002, pages D-12 to 13).

In total, the primary requirements for flycatcher habitat are hydrologic scouring events that cause the riparian habitat to be recycled through disturbance and a water supply or table close enough to the surface to support the dense riparian vegetation used to establish and maintain nesting territories. Willows need a consistent supply of soil water to survive and this condition is likely to be either close to the river where shallow groundwater and capillary action keeps soils moist or in locations susceptible to overbank flooding during the growing season (Section 3.3.1). Because of this, the apparent need for overbank and below-nest flooding is mostly related to maintaining dense stands of vegetation for flycatcher nesting habitat than a specific requirement for nesting success. While it may not be a requirement for maintaining successful nest sites, surface water occurring under or near potential nest sites may be a criterion promoting the initial establishment of riparian vegetation. Additional investigation is necessary to clarify the surface water requirements of the flycatcher.

3.2.4.3 Intermittent Flows and Nesting Success

The 2003 BO reports that "in 1996, at least 36 river miles in the MRG were dry for 128 days. This event may have contributed to complete failure of adjacent flycatcher nests (Johnson et al. 1999)." It also reports that "flycatchers may also abandon sites if they are dry in May and early June, during the birds' pair bonding and early nesting chronology (Johnson et al. 1999)." Because of the important implications of these statements and their apparent pivotal use in the BO and development of the RPAs, it is useful to recognize that the study was based on only 9 territories near San Marcial and 5 territories near Velarde during 1996. At San Marcial, 5 of 9 male flycatchers remained unpaired during both 1996 and 1997, accounting for the failure of those 5 "nests." Thus, these failures were related to a lack of females in years with and without flow. Range-wide, the occurrence of unpaired males is not uncommon, with the skewed sex ratio possibly being related to increased female mortality (Stoleson et al., 2000).

With regards to the remaining 4 pairs at San Marcial in 1996, one nest was found and it showed evidence of failure associated with predation and most likely parasitism. The fate of the remaining 3 pairs is uncertain, although they could have nested in another area. Observations from San Marcial in other years indicate that not all paired flycatchers nest in the same territories year after year, whether the territory is flooded or not (Moore and Ahlers, 2003). Sogge (2000) indicated that flycatchers display nest-site fidelity, although banding studies in Arizona revealed a return rate of about only 30 percent. About 10 percent of the first year birds moved to new areas, and the fate of the remaining 60 percent was undetermined.

Other factors besides flow can affect nesting success. For instance, none of the 5 territories near Velarde studied by Johnson et al. (1999) resulted in a successful nest in either 1996 or 1997, despite the presence of continuous flows near the territories in both years. Thus, the relationships between successful nesting and intermittent flows are not well understood.

3.2.4.4 Habitat Suitability Model

Reclamation developed a flycatcher habitat suitability model in 1998 that continues to be refined and applied between Bernardo and Elephant Butte Reservoir (Ahlers and White, 2001; Moore and Ahlers, 2003). This model uses the vegetation classes developed by Hink and Ohmart (1984), with recent modifications to account for distance to surface water. With regards to breeding habitat suitability the model was refined using appropriate vegetation units and whether the vegetation units are within 100 m (330 feet) of existing watercourses, ponded water, or a zone of peak inundation. The 5 categories of suitability are as follows:

1. **Highly Suitable Native Riparian** – Stands dominated by willow and cottonwood, having adequate structure with a dense understory and within 100 m of water.
2. **Suitable Mixed Native/Non-native Riparian** – Includes stands of native mixed with various compositions of non-native species and within 100 m of water.
3. **Marginally Suitable Non-native Riparian** – Stands composed of monotypic saltcedar or stands of saltcedar mixed with Russian olive within 100 m of water.
4. **Potential with Future Riparian Vegetation Growth and Development** – Includes stands of very young sparse riparian plants, river bars, and open younger cottonwood stands that could develop into stands of adequate structure and within 100 m of water.
5. **Low Suitability** – Includes areas more than 100 meters from water or where native and/ or non-native vegetation lacks the structure and density to support breeding flycatchers, areas dominated by young mixed cottonwood-saltcedar stands, flooded dead saltcedar stands, open water areas (rivers, canals, lakes), and roads.

Currently, the Service groups the first three categories as equally suitable because nesting sites occur in these three vegetation communities (FWS, 2003a). Reclamation indicates that they have documented flycatcher territories only in the Highly Suitable Native Riparian group, although saltcedar is always a structural component in the areas where flycatchers occur (see Table 1 in Moore and Ahlers, 2003).

To date, the habitat mapping has only been applied to areas south of Bernardo through the San Marcial reach and into the upper end of the Elephant Butte Delta. Based on this model, 9,998 acres (4,046 ha) of highly suitable, suitable, and marginally suitable habitat; 4,478 acres (1,812 ha) of potentially suitable habitat; and 47,190 acres (19,091 ha) of low suitability habitat exist in their study area. If one conservatively assumes that each of 147 territories mapped south of Bernardo occupies 1 acre (0.4 ha), then only 7 percent of the presently available highly suitable habitat under Reclamation's definition and only 1.5 percent of the suitable habitat under the FWS definition is now being used along that reach. This relationship is consistent with the findings from several other studies showing areas containing apparently highly suitable nesting habitat that lacked nests even though non-nesting individuals occupied the site (FWS, 2002). Thus, either surplus habitat exists in the MRG or there are factors that affect nesting but have not been identified. Focus should be given to better understanding recruitment factors for the flycatcher.

3.2.4.5 Critical Habitat Linkages

The strongest link between water supply and flycatcher breeding success appears related to the production of dense riparian vegetation. As indicated above, the water requirements for flycatcher breeding habitat would include flooding to establish willows and a water table close enough to the surface to maintain the riparian vegetation used for nesting. Historically, large scouring floods created spatially distinct riparian communities with uneven age structures. In contemporary times, however, destructive flood events have been controlled and saltcedar and other introduced species compete with willows along much of the Rio Grande (Section 3.3).

The importance of saltcedar in maintaining flycatcher habitat is controversial. Unlike the willow species, saltcedar maintains a suitable twig structure for flycatcher nest sites as the trees mature. As water management and bank protection removed the effects of floods from the Rio Grande system, a working hypothesis exists that the replacement of willow by saltcedar may have helped maintain flycatcher populations along the MRG by providing needed nesting structures for flycatcher at the time that aging willows lost the capacity to provide suitable nest sites.

The Service cautions that "throughout the western U.S., large tracts of tamarisks [saltcedar] are being cleared for purposes including water salvage, flood water conveyance, and/or wetland [riparian] restoration. Such actions pose a threat to southwestern willow flycatchers when conducted in areas of suitable habitat (occupied or unoccupied) and when conducted in the absence of restoration plans to ensure replacement by vegetation of equal or higher functional values" (FWS, 2002, page 39).

3.3 The Riparian Forest Ecosystem of the Rio Grande

Large-stature cottonwoods and their understory cohorts define the character of the Rio Grande bosque and many other riparian areas in the southwestern United States. In the MRG, the vegetation density in the bosque strongly contrasts with the surrounding uplands, which are sparsely vegetated reflecting the inherent aridity of the region. Table 3-2 lists the native and non-native species that predominate in the riparian plant communities in the MRG today.

Depth to groundwater is the primary determinant of the composition and distribution of riparian plant communities across the floodplain (Muldavin et al., 2000; Crawford et al., 1993). Cottonwoods and willows occur in those areas where a shallow water table fluctuates seasonally but rarely stays near the surface for extended periods. Understory species in the bosque are a function of latitude and stand age in addition to groundwater depth (Muldavin et al. 2000; Dick-Peddie, 1993). Areas of poorly drained soils that experience brief and infrequent flooding and where the groundwater is near the surface (< 2 feet) have the potential to translocate and accumulate salts in the upper soil horizons, thereby forming saline-alkali meadows (Muldavin et al. 2000). The resulting plant community is typified by saltgrass and other salt-tolerant species like alkali sacaton, saltbush, and rabbitbush.

Wetlands and marshes form on the floodplain in geomorphic positions (i.e. topographic depressions or abandoned channels) that intersect the water table, creating permanent or semi-permanent ponded conditions. Persistent herbaceous wetlands in the MRG also occur in areas that are seasonally flooded and have saturated soils within 2 feet of the surface (Muldavin et al. 2000). Herbaceous wetlands contain numerous graminiods (e.g., cattails, sedges and rushes) and may have a diverse array of forbs depending on soil texture and chemistry, hydrologic regime, and geomorphic setting. Alkali meadows and wetlands were more widespread in the MRG prior to improvements to irrigation systems beginning in the 1930s (Wozniak, 1998). Prior to that time, local irrigation systems were open-ended, dumping water on the alluvial terraces at the end of ditches rather than returning it to the river. Flow reductions and sedimentation raised the groundwater table and conspired with these poor irrigation practices to increase salinity and drainage problems throughout the valley (U.S. Reclamation Service, 1922).

Table 3-2: Representative riparian plant species in the MRG

Bosque	
Common Name	**Scientific Name**
coyote willow	*Salix exigua*
Goodding's willow	*Salix gooddingi*
Baccharis	*Baccharis wrightii*
New Mexico olive	*Foresteria neomexicana*
Sumac	*Rhus trilobata*
screwbean mesquite	*Prosopis pubescens*
wolfberry	*Lycium andersonii*
false indigo bush	*Amorpha fruticosa*
senna	*Cassia bauhinioides*
ground rush	*Juncus balticus*
Rio Grande cottonwood	*Populus deltoides* ssp. *wislizeni*

Alkali Meadow	
Common Name	**Scientific Name**
saltgrass	*Distichlis stricta*
alkali sacaton	*Sporobolus airoides*
saltbush	*Atriplex* spp.
rabbitbush	*Chrysothamnus* spp.

Wetland	
Common Name	**Scientific Name**
cattail	*Typha* spp.
willow	*Salix* spp.
sedge	*Carex* spp
rush	*Juncus* spp.
reed canarygrass	*Phalaris arundinacea*
horsetail	*Equisetum* spp.
buttercup	*Ranunculus cymbalaria*
yerba-mansa	*Anemopsis californica*

Non-native Species	
Common Name	**Scientific Name**
saltcedar	*Tamarix* spp.
Russian olive	*Elaeagnus angustifolia*
Siberian elm	*Ulmus pumila*
tree of heaven	*Ailanthus altissima*
white mulberry	*Morus alba*

3.3.1 Ecology of Cottonwood and Willows

Much of the research on southwestern riparian communities has focused on the ecology of cottonwoods, with an emphasis on conditions that promote their maintenance and regeneration. Unfortunately, technical information on willow ecology is not as comprehensive, although their requirements are quite similar to their cousin the cottonwood.

The willow family (*Salicaceae*), represented by cottonwoods (*Populus species*) and willows (*Salix species*), is prevalent in riparian ecosystems throughout the western United States. The taxonomy of cottonwood is somewhat complicated by natural variation and hybridization among sympatric species (Wyckoff and Zasada 2003; Zasada et al. 2003). In the case of the Rio Grande cottonwood, scientific and common nomenclature is inconsistent among authors (Lamb 1989; Carter 1997; Dick-Peddie 1993; Sher et al., 2000), though Eckenwalder's (1977) treatise on the genus *Populus* identifies the cottonwood in the MRG as *P. deltoides* ssp. *wislizeni*,. Narrowleaf cottonwood (*P. angustifolia*) occurs further north in the Rio Chama and Rio Grande gorge above Velarde and in tributary drainages above 6,500 feet (Dick-Peddie 1993). Cottonwoods do not have mechanisms to isolate themselves genetically, and as a result, they hybridize where the species ranges overlap (Braatne, 1999). Fremont cottonwood has been known to hybridize not only with narrowleaf cottonwood (Taylor, 2000), but also with plains cottonwood (Braatne, 1999).

The life histories and ecological requirements of cottonwoods are fairly consistent throughout their range (Alberta, Canada to New Mexico and California). Rio Grande cottonwood is a fast-growing tree (Taylor, 2000; Sher et al., 2000) that can live for 130 years or more and attain 12 to 35 m in height. Cottonwoods are dioecious plants, producing wind-pollinated flowers at the age of 5 to 10 years (Gladwin and Roelle, 1998). The seed is both wind- or water-borne and shade-intolerant, requiring an open, moist seedbed for germination. In the MRG, seeds are dispersed in early to late June, depending on latitude. Localized differences in seed release dates should be considered in interpreting results of studies conducted in other regions (Taylor, 2000). Asexual reproductive responses (suckering) of the Rio Grande cottonwood associated with to minor injury due to mechanical damage or flooding are relatively uncommon, especially as trees become older (Taylor, 2000 and 2001).

Cottonwood is a facultative riparian plant, preferring to have its roots within the capillary influences of the water table during the growing season (Busch and Smith, 1995; Stromberg et al., 1991). In alluvial river valleys, cottonwood forests on the floodplain depend primarily the shallow groundwater that is recharged by in-stream flows (Stromberg and Patten, 1996). The persistence of cottonwood in the Southwest has been linked to sites with water tables within 3 m of the soil surfaces, although mature cottonwoods may persist with groundwater as deep as 7 to 9 m (Lacey et al. 1975; Reichenbacher 1984; Stromberg et al. 1996; Scott et al., 1999). Cottonwood water use, however, is not solely limited to groundwater sources; and cottonwood has the ability to use precipitation stored in upper soil layers during the growing season, particularly in ephemeral systems (Snyder and William, 2000).

Common willow species in the MRG include Goodding's (*S. gooddingii*) in the south and peachleaf (*S. amygdaloides*) and coyote (*S. exigua*), which increase in frequency in the northern reaches (Muldavin et al., 2000). Most willows are rapidly growing shrubs or small multi-stemmed trees that are relatively short-lived. In the MRG, they typically occur in dense thickets in sites immediately adjacent to water and with water tables within 1.5 m of the soil surface (Muldavin et al., 2000). Once established, willows have a very high flood tolerance, leading to the stabilization of channel banks. Stromberg (1997) noted that Goodding's willow was more successful on wetter sites nearer the channel compared to cottonwood. The willow's adaptation to bankline habitats prone to scour is to develop lateral (spreading) roots at the expense of deep roots (Horton and Clark, 2001). Thus, rapid declines in the water table can produce significant die-backs of willow saplings (Shafroth et al. 2000; Horton and Clark, 2001). Because most species are shade-intolerant, shorter species like coyote willow are replaced by larger trees over time as the canopy expands. Goodding's willow, however, grows much taller than coyote willow (8 to 30 m) and frequently is a codominant with cottonwood (Reed, 1993).

Willow flowers are borne in catkins and are pollinated primarily by bees. Willows regenerate through the dispersal of thousands of small seeds by wind or water (Reed, 1993; Uchytil, 1989a and b). Reed (1993) noted that Goodding's willow seedlings do not compete well with grasses and won't sprout beneath the willow's own canopy. Most willows will also regenerate readily from stem cuttings and the crown, coyote willow will clone itself from root sprouts (Uchytil, 1989a).

The life histories of cottonwoods and willows have evolved to take advantage of two types of flooding. First, episodic floods of moderate to large magnitude create bare soil areas that served as recruitment sites. Being shade-intolerant, cottonwood and willow seeds require an exposed mineral soil seedbed to germinate. Second, spring overbank flooding prepared a seedbed by removing litter and bringing the soils to field capacity. It is unlikely that the more predictable spring overbanking in the MRG assisted with seed dispersal as snow melt flows would have receded by mid- to late-June (Section 3.3.4.4). Elongating roots of newly germinated seeds probably depended on stored soil water in the upper soil horizons until they encountered the capillary zone above the water table. If antecedent soil water was exhausted prior to the roots reaching groundwater, the saplings probably would not survive without additional water as precipitation or runon. A more complete discussion of the fluvial-geomorphic requirements for cottonwood recruitment is provided Section 3.3.4.

3.3.2 Ecology of Woody Exotic Species

The primary large stature weed species in the MRG include saltcedar, Russian olive (*Elaeagnus angustifolia*), and Siberian elm (*Ulmus pumila*). These plants were introduced to the Southwest in the late 1800s and early 1900s for shelterbelt plantings, slope stabilization, erosion control, and landscaping.

Saltcedar was first brought to the United States in 1837 (Stenquist and Kauffeld, 2000) and plant nurseries began to sell saltcedar as early as the 1850s (Brotherson and Winkel, 1986). Wooton and Standley (1915) reported that saltcedar often escaped cultivation in New Mexico. Saltcedar was also used extensively for erosion control along the Rio Puerco and Galisteo River in the early 1900s (MacDonald, 1955; Everitt, 1998). Other introduced trees that are becoming part of the bosque, particularly in urbanized areas, are white mulberry (*Morus alba*) and tree of heaven (*Ailanthus altissima*) (Crawford et al., 1993).

Saltcedar is an aggressive competitor with multiple modes of reproduction and a broad ecological amplitude. Reproductively, it has an edge over native species because a single plant can produce 500,000 seeds per year (Stenquist and Kauffeld, 2000) from mid June to September, allowing colonization of open sites when summer precipitation affects soil moisture conditions (Brotherson and Winkel, 1986; Warren and Turner, 1975). Flowers may be self- or insect pollinated and seeds are transported by wind and water and remain viable for about 2 months (Horton et al., 1960; Zouhar, 2003). If conditions are right, seeds can germinate 24 hours after dispersal, sometimes while still floating on water (Shepperd, 2003). Saltcedar can also root sprout and layer from detached twigs buried in sediment (Shepperd, 2003). Saltcedar matures and produce seeds the year following establishment (Gladwin and Roelle, 1998).

Saltcedar can tolerate inundation and drought, and resprouts vigorously following fire and mechanical disturbance (Warren and Turner, 1975; Stevens and Waring, 1985; Stromberg, 1998a). Compared to cottonwood, established saltcedar is more tolerant of groundwater declines and can persist in the absence of a permanent water table (Busch and Smith, 1995; Shepperd, 2003, Horton et al., 2001). In addition, saltcedar can withstand longer periods of inundation than cottonwood (Warren and Turner, 1975; Sprenger et al., 2001).

Russian olive is native to Eastern Europe and Central Asia and was introduced to New Mexico in the early 1900s (Hink and Ohmart, 1984). This small, drought-tolerant tree produces large seeds that are dispersed by birds. Russian olive is tolerant of both shade and full sunlight, and also fixes nitrogen. It can therefore colonize new disturbances or encroach on established stands of cottonwoods.

Siberian elm was introduced to the United States from northern China in 1860 as a shade tree (Natural Resources Conservation Service [NRCS], 2002). Siberian elm is shade- and drought-tolerant and produces a wind-borne, winged fruit (samara) that maintains viability for many years. Siberian elm sprouts from both its root and crown. It is increasingly represented along several reaches of the MRG, including areas around Albuquerque. Its prevalence in the Albuquerque area is probably related to the availability of seed sources in the surrounding areas.

3.3.3 Ecosystem Dynamics of the Middle Rio Grande Bosque

The failure of native species like cottonwood to dominate riparian areas is viewed as a symptom of the altered hydrologic regime of managed rivers (Johnson et al., 1976; Rood and Mahoney, 1990; Smith et al., 1991; Fenner et al., 1985; Shafroth et al., 2000). Changes in hydrology (i.e., decreased frequency and magnitude of periodic and episodic flood events, alteration of the alluvial water table) and geomorphology (i.e., less channel migration) have been implicated as causing the decline of the native riparian forest, including the lack of regeneration necessary to diversify tree age classes (Howe and Knopf, 1991), the increase in exotic woody species (Busch and Smith, 1995), and the alteration of the alluvial water table such that it either stresses and kills mature plants (Scott et al., 1999) or fails to support seedlings (Mahoney and Rood, 1998).

A decline in cottonwood recruitment (Howe and Knopf, 1991) and an increase in the age structure of cottonwood forests (Mount et al., 1996) have been observed in the MRG. The density of saltcedar and Russian olive has steadily increased throughout the Isleta-Belen reach (Mount et al., 1996). Exotic woody species represent a large component of new riparian communities on islands and bars in the Albuquerque reach (Milford et al., 2003). However, the proliferation of saltcedar along unregulated streams suggests that flow regulation is not the only factor affecting riparian community structure and saltcedar's dominance. Saltcedar has many mechanisms that give it a competitive advantage: it survives and grows on both the wet and dry sites (Stromberg 1997), it withstands more dramatic groundwater declines than native species (Shafroth et al., 2000; Scott et al., 1999), and it's generally unpalatable to livestock and has longer seeding periods (Zouhar, 2003).

In some areas of the country, rapid declines in water tables have produced significant die-backs of mature cottonwood (Scott et al., 1999) and cottonwood saplings (Shafroth et al., 2000). However, established individuals can tolerate slow declines in water tables. Because the channel and riverside drains control the shallow groundwater table in the MRG, it is unlikely that declines in groundwater contribute to a loss of cottonwoods. Horton et al. (2001) found that regulated flows may actually benefit cottonwood and willow by maintaining a relatively constant groundwater depth as opposed to an unregulated stream with greater water table fluctuations where natives experienced greater physiological stress.

Stromberg (1998a) compared the structural and functional equivalency of saltcedar to cottonwood in a free-flowing riparian system. Similarity was assessed by comparing the means and temporal trends (changes with stand age) of stands dominated either by saltcedar or cottonwood in Arizona. She found that the two species were functionally equivalent for about half the traits examined. However, many of the parameters construed as indicators of riparian ecosystem function (e.g., soil particle size fractions, sedimentation rates and distance to channel) may be characteristics inherent to a site rather than dependent variables affected by the tree canopy. With regards to those properties that could be affected directly by the overstory component, saltcedar stands had significantly more understory cover in both the spring and fall and higher spring floristic diversity than cottonwoods. Conversely, older stands of cottonwoods had significantly more litter than saltcedar stands, which may account for the reduced herbaceous cover in the understory. Structurally, cottonwood stands had greater basal area and canopy height than saltcedar. Because many of the traits did not differ between the two species, Stromberg (1998a) concluded that the functional role of saltcedar is context-specific and variable among rivers.

3.3.3.1 Edaphic Conditions

Salinity: Soil salinity may affect the establishment and survival of riparian species (Busch and Smith, 1995; Pinkney, 1992). Salinity is primarily a concern during the germination phase, but is less important for mature plants (Rhoades et al., 1992). Similarly, salinity levels in the surface layers are a greater concern than deeper in the soil profile.

Salinity tolerance levels for mature riparian vegetation have not been established for the Middle Rio Grande; however all but the most sensitive plants can tolerate salinity levels up to 4 dS/m with only marginal reductions in productivity (Rhoades, 1992). Additionally, Campbell and Dick-Peddie (1964) found no correlation between total soluble salts and established stands of dominant riparian species along the Rio Grande.

Siegle and Brock (1990) reported significant reductions in germination for Goodding's willow at salinity levels above 10 dS/m (100 meq/l NaCl), and for Fremont cottonwood above 5 dS/m (50 meq/l NaCl). Glenn et al. (1998) reported only minor reductions in relative growth rates of willow and cottonwood saplings at salinity levels less than 12.5 dS/m (8 g/l NaCl). Saltcedar saplings tolerated salinity levels of 50 dS/m (32 g/l NaCl) with only minimal declines in relative growth rates. Chloride salts were used in the previous studies to avoid the confounding effects of solubility limitations associated with sulfate salts. However, none of these studies addressed the potential for specific ion toxicity associated with chloride as opposed to osmotic effects.

Rio Grande waters tend to be dominated more by sulfates than chlorides. Using a mixed salt solution to better emulate Rio Grande water, Shafroth et al. (1995) found no negative effects on cottonwood germination with salinity levels up to 4.5 dS/m (about 7x river concentration). They concluded that salinity was a minor factor in regulating germination and growth of saltcedar and cottonwood on Rio Grande sand bars. Swenson and Mullins (1985) reported that cottonwood and willow pole plantings failed where groundwater and soil salinity levels exceeded 6.25 dS/m on Pecos River sites.

The highest soil salinity levels generally occur in association with evapoconcentration processes in areas where the capillary fringe extends to the soil surface. Saltgrass typically dominates such sites since it can tolerate soils with salinity levels greater than 5 dS/m (USFS 2003). In the MRG, saltgrass was associated with areas where salts had accumulated and where the water table was within 4 feet of the surface (Campbell and Dick-Peddie 1964). Figure 3-8 shows the soil salinity levels in established saltcedar and cottonwood stands in Arizona and New Mexico. These data indicate that both species can grow within a similar range of soil conditions, although the highest values were found in a saltcedar stand.

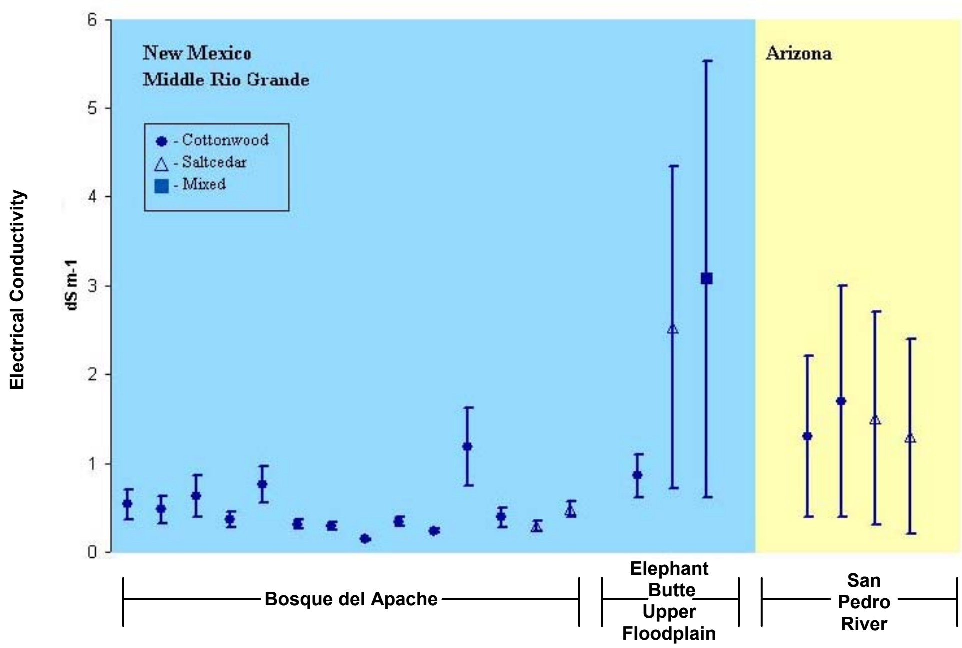

Figure 3-8: Soil electrical conductivity values for mature cottonwood and saltcedar stands.
(Adapted from Stromberg, 1998a; BOR, 1975; Ellis et al., 1996)

Soil Texture: Generally, saltcedar favors finer textured soils compared to cottonwoods, which often pioneer coarse-textured alluvial materials (Stromberg et al., 1991; Stomberg, 1997 and 1998a). Willows are more often found in, but not limited to, finer textured soils (Johnson et al., 1976; McBride and Strahan, 1984).

3.3.4 Riparian Vegetation and Fluvial-Geomorphic Relationships

The life cycle of native species in the Rio Grande bosque evolved with and is intricately linked to the temporal dynamics of the river (Scott et al., 1997). The structure, function, and composition of riparian ecosystems are dependent upon episodic disturbances such as those caused by infrequent high flows (Rood and Mahoney, 1990; Busch and Scott, 1995; Friedman et al,. 1997; Osterkamp, 1999). Riparian plants that are associated with flycatcher habitat specifically, and the riparian forest in general, are well adapted to episodic and periodic flooding. In general, riparian plants share a number of physiological and life-history attributes that allow new age classes to develop within a riparian ecosystem; these attributes are large annual seed crops, early spring bloom, early summer seed dispersal, wind- and water-dispersed seed, short seed viability, low shade tolerance, high germination rates, and vegetative reproduction.

In the MRG, flood control, channelization, channel degradation, and recent climatic conditions have reduced the frequency of disruptive flood events that originally enhanced the potential for intermittent replacement of riparian vegetation. Overbank flooding coincident with spring runoff has been proposed as a possible mechanism for regenerating cottonwood and willow communities in the MRG (FWS, 2003a; Sites Southwest, 2002; Crawford et al., 1999; Ellis et al., 1996). Return periods for bankfull flows typically range from 1.5 to 3 years (NRC, 2002). Overbank flows for the Rio Grande at the Albuquerque gage are defined here as those with a return interval of 1.5 to 2 years of approximately 5,000 cfs.

Feeding and nesting habitats for the migratory, riparian-obligate flycatchers were presumably established and maintained historically by natural flood regimes. Many researchers have discussed the role of flooding in riparian vegetation establishment (e.g., Scott et al., 1997; Stromberg et al., 1991; Sparks, 1995; Bradley and Smith, 1996). However, if flooding is to be a tool to restore flycatcher habitat, the magnitude, frequency, timing, and duration of flooding required to regenerate native riparian plant species must be carefully defined. Moreover, the role of periodic overbank flooding today as a means to regenerate flycatcher habitat needs to be carefully examined. The following subsections discuss the interrelationships among the components of flow, specifically relative to its potential use to restore native riparian vegetation to benefit the flycatcher.

3.3.4.1 Frequency

Flow frequency relates to how often a flow of a certain magnitude is equaled or exceeded in a given time interval (NRC, 2002). Flow magnitude in intricately linked with frequency, providing a measure of the available energy the river has to do work (NRC, 2002). To open new areas for cottonwood and willow establishment, the river must have sufficient energy clear old vegetation, erode banklines, and deposit fresh sediment above the active channel. Stromberg et al. (1991) determined that cottonwood and willow recruitment took place at an average recurrence interval of about 12 years on the Hassayampa River in Arizona. Cottonwood recruitment on point bars of a meandering river in North Dakota occurred once every 5 years in response to medium to high flows (Bradley and Smith, 1986). Bradley et al. (1991 in Hughes, 1994) estimated that 30- to 50-year recurrence interval flood were required for cottonwood/willow establishment on streams in southern Alberta. Mahoney and Rood (1998) estimated that soil moisture and stage conditions in areas with exposed soils were right for cottonwood establishment about 1 in every 5 to 10 years. Moreover, successful recruitment typically requires a high-flow event followed by at least 1 year of subsequently lower intensity flooding to promote seedling survival (Auble, 1999; Mahoney and Rood, 1998).

The exact frequency of moderate- to high-magnitude flows required to initiate new riparian plant communities in the Rio Grande bosque has not been established. Figure 3-9 indicates that historically, several years may pass with no significant flow events resulting in floodplain inundation in the Middle Valley. Moderately high flows (>5,000 cfs) occur at irregular intervals at the Albuquerque gage (Fig. 3-

9). Several high magnitude floods with return intervals of 50 and 100 years occurred in the southern reaches during a 12-year period (MEI, 2002; Happ, 1948; MacDonald, 1955), scouring new channels while abandoning older channels. It is possible that the flood stage that significantly modifies the channel has not been measured in the northern reaches since the gages were installed, despite several flows above 10,000 cfs prior to the construction of Cochiti Dam. Given the current safety, legal, and climatic restraints on the system, such flows are not likely to occur in the future.

Frequent flooding may limit cottonwood and willow regeneration. Recruitment is limited in fresh sediment in low-lying areas because seedlings established in the spring may be removed by subsequent summer floods or the succeeding year's peak flows (Scott et al., 1993; Gladwin and Roelle, 1998; Stromberg et al., 1991). Across the geographical distribution of cottonwood, seedlings are successful when they become established 0.6 to 2 m above base flow, where they can escape periodic flood events (Scott et al., 1993; Stromberg, 1997; Mahoney and Rood, 1998). Conversely, the use of late -season floods has been suggested as a means to control saltcedar seedlings in the channel (Gladwin and Roelle, 1998).

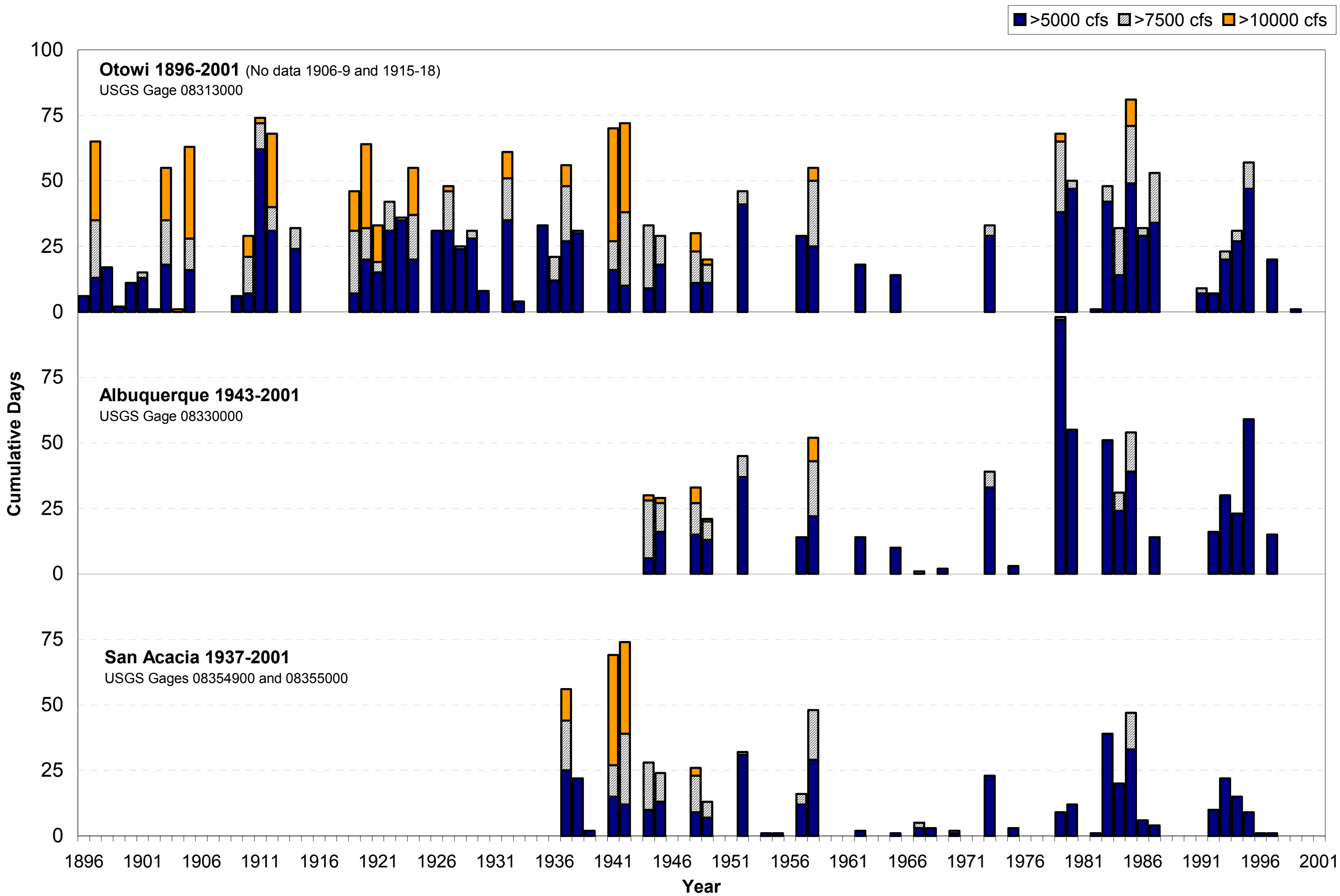

Figure 3-9: Flow frequency and duration for the Rio Grande at three gages

3.3.4.2 Magnitude

Magnitude refers to the amount of water flowing past a given location during a unit of time and is a critical variable in the creation of local zones of erosion and deposition for recruitment (NRC, 2002). Infrequent high-magnitude floods that widen the channel, scour vegetated banks, create new avulsions or deposit large amounts of sediment initiate the recruitment process (Friedman et al., 1997). Point, alternate, and medial bars provide additional sites for colonization during periods with sequential years of low to moderate (non-scouring) flow.

Scott et al. (1997) indicated that flows of 1,400 cms (~ 49,000 cfs) were sufficient to destroy existing floodplain plant communities and provide sites for cottonwood regeneration in the upper Missouri River basin. The persistence of the in-channel stands depended on the size of subsequent flood events and degree of vegetation establishment. Scott et al. (1996, 1997) indicated that channel narrowing associated with riparian expansion occurs more often on braided semi-arid rivers with sand and gravel beds and that have large variations in width and flow. Riparian forest expansion (channel narrowing) during periods of low flow has been documented on the Rio Grande (Williams and Wolman, 1984).

The magnitude of flows needed to destroy established vegetation and expose germination sites along the Rio Grande is unknown. Historically, the abandonment of channels and creation of new channels during avulsions created sites for colonization along the Rio Grande. More recently, colonization of medial and lateral bars has become prevalent along the Rio Grande following the cessation of channel clearing operations and the relatively long and persistent high-stage flows of the mid 1980s, followed by the low-water years of the late 1990s.

A flow with a magnitude of 5,000 cfs on the Rio Grande is expected to advance onto the floodplain in some reaches, but not dramatically alter the vegetation or cause bank erosion. Though the post-Cochiti Dam frequencies will vary by reach, the approximate return period for a 5,000 cfs flow is 2 years. Overbank flooding has been observed at flows that exceeded 5,000 cfs in the Albuquerque, Isleta, San Acacia and San Marcial reaches (Crawford et al., 1993). Notably, moderate to high flows were fully contained in the channel between Cochiti and Angostura (7,500 cfs) and between Bernardo and San Acacia (5,400 cfs) (Crawford et al., 1993). Overbank flooding does occur at lower flows (3,000 cfs) in some areas of the San Acacia reach.

3.3.4.3 Duration

The period of time associated with a specific flow magnitude defines flow duration (NRC, 2002). Flow duration may effect the establishment of riparian vegetation by controlling the rate at which the water table recedes (Taylor et al., 1999). Cottonwood and willow require adequate available soil water in the upper root zone to allow the seedling roots to elongate to the capillary fringe. If the antecedent water content and storage capacity of the soils is inadequate, and soil water is augmented by a stage-controlled groundwater recharge from the river, then a short-duration, quickly-receding flood may affect seedling survival. Conversely, extended flooding may prevent germination and reduce survival if water logged conditions persist for too long. Long-duration floods in the late summer and fall could also contribute to native seedling and sapling mortality or the promotion of saltcedar.

Mahoney and Rood (1998) indicated that the roots of cottonwood seedlings grow about 0.5 to 1 cm per day and rapid river stage declines may impact their survival. Taylor et al. (1999) reported that the survival of cottonwood, coyote willow, and saltcedar was improved when the river stage declined at slower rates compared to the preceding year when the declines were more rapid. There is an inverse relationship between the rate of drawdown and the survival of cottonwood (Sprenger et al., 2002) and Goodding's willow seedlings (Horton and Clark, 2001). Summer rains are likely to augment the soil water available to seedlings, especially those that germinate later in the season. Relatively high mortality rates are expected in the first few years for sites with marginal soil water conditions.

Flow-duration curves are commonly used to describe the probability of sustained flows of a given size. However, flow-duration curves provide little information on the actual sequencing of flows and are therefore of limited use in assessing the effects of flood duration on riparian vegetation communities. For example, on average flows greater than 5,000 cfs at the Albuquerque gage for the period of 1943 to 1999 occurred about 5 percent of the time (MEI, 2002). Biological implications cannot be surmised from flow-duration or flood-frequency probability curves because they fail to capture the annual and seasonal variability of flood events. The cumulative durations of higher magnitude flows at three gages along the Rio Grande are shown on Figure 3-9. These figures indicate that the duration of flows with the potential to cause overbank flooding is quite variable, ranging from just a few days to several months. The effects of Cochiti Dam and the irrigation diversions are evident; the Otowi gage frequently measured larger flows of longer duration than those measured downstream. Whether stage-controlled groundwater declines are an important factor in native species recruitment in the MRG is unknown, however. Floodplain soils in the MRG are not excessively coarse and could contain adequate water to support small saplings until their roots reach the capillary zone of the water table. Moreover, channel and riverside drains control the absolute level of the shallow groundwater table in the MRG, and it is unlikely that severe declines in groundwater contribute to a loss of mature cottonwoods.

3.3.4.4 Timing

Flow timing refers to the seasonality of a given flow (NRC, 2002). The timing of flood events was less crucial to native plant recruitment before exotics were present. Prior to saltcedar introduction, cottonwoods and willows could have become established any number of years following a disturbance (e.g., floods or fire) when conditions were right. Today, colonization sites can be occupied by saltcedar and other exotic species, which have longer seed dispersal and viability periods than native species (Table 3-3).

Annual peak flows that result in overbank flooding in the MRG don't always coincide with native seed dispersal and viability periods (Fig. 3-10). The competitive advantage of the non-native species is their longer seed dispersal and viability periods. In a study of riparian communities between Albuquerque and El Paso, Campbell and Dick-Peddie (1964) indicated that areas flooded during the growing season had higher saltcedar densities than areas flooded in the early spring and with deeper water tables. Figure 3-10 demonstrates that, in most years, overbank flows due to snowmelt end by mid to late May, which may preced seed dispersal in some areas. Though cottonwood and willow seeds are dispersed by water, their dependence on water dispersal for placement on a suitable floodplain site is limited to those years when spring flows occur after seed dispersal. For cottonwood, seed dispersal generally occurs in June and July in the Albuquerque area (Wykoff and Zasada, 2003) and between late May and early June in the Bosque del Apache area (Gina Dello Russo, personal communication). Seed dispersal occurs in June and July for Gooddings willow (Zasada et al., 2003). These observations establish the importance of correct timing of flooding and duration of flows whenever flow manipulation is to be included as a restoration technique.

Table 3-3: Reproductive attributes of selected MRG vegetation

		MRG Timing					
Species	**Initial Seed-Bearing Age**	**Flowering**	**Seed Dispersal**	**Seed Viability[a]**	**Seeds/kg**	**Vegetative Propagation**	**References**
Populus deltoides	5-10 years	April-May	June-July	1-5 weeks	770,000	roots, layering (weak)	Wyckoff and Zasada, 2003; Siegel and Brock, 1990; Taylor, 2001
Salix gooddingii	2-10 years	May-June	June-July	1 week	ND	crown, layering	Zasada et al., 2003; Siegel and Brock, 1990: Reed, 1993
Salix amygdaloides	ND	May-June	June-July	1 week	6,420,000	crown, layering	Uchytil, 1989b
Salix exigua	ND	May-June	June-July	1 week	22,000,000[b]	crown, roots, layering	Zasada et al., 2003; Pratt et al., 2003; Uchytil, 1989a
Baccharis salicifolia	ND	May-July	Sept-Oct	2 years	11,000,000[b]	none	Karrflat and Olson, 2003
Tamarix ramosissima	1-3 years	mid-May-Sept	June-Sept	45 days	ND	roots, crown, layering	Zouher, 2003; Horton et al., 1960
Ulmus pumila	ND	March-April	April-May	8 years	158,000[b]	crown, roots	Barbour and Brinkman, 2003
Elaeagnus angustifolia	3-5 years	June	Sept-Mar	3 years	11,380[b]	crown, roots	Olsen and Barbour, 2003; Teskey, 1992

ND - no data
[a] under field conditions
[b] cleaned seeds

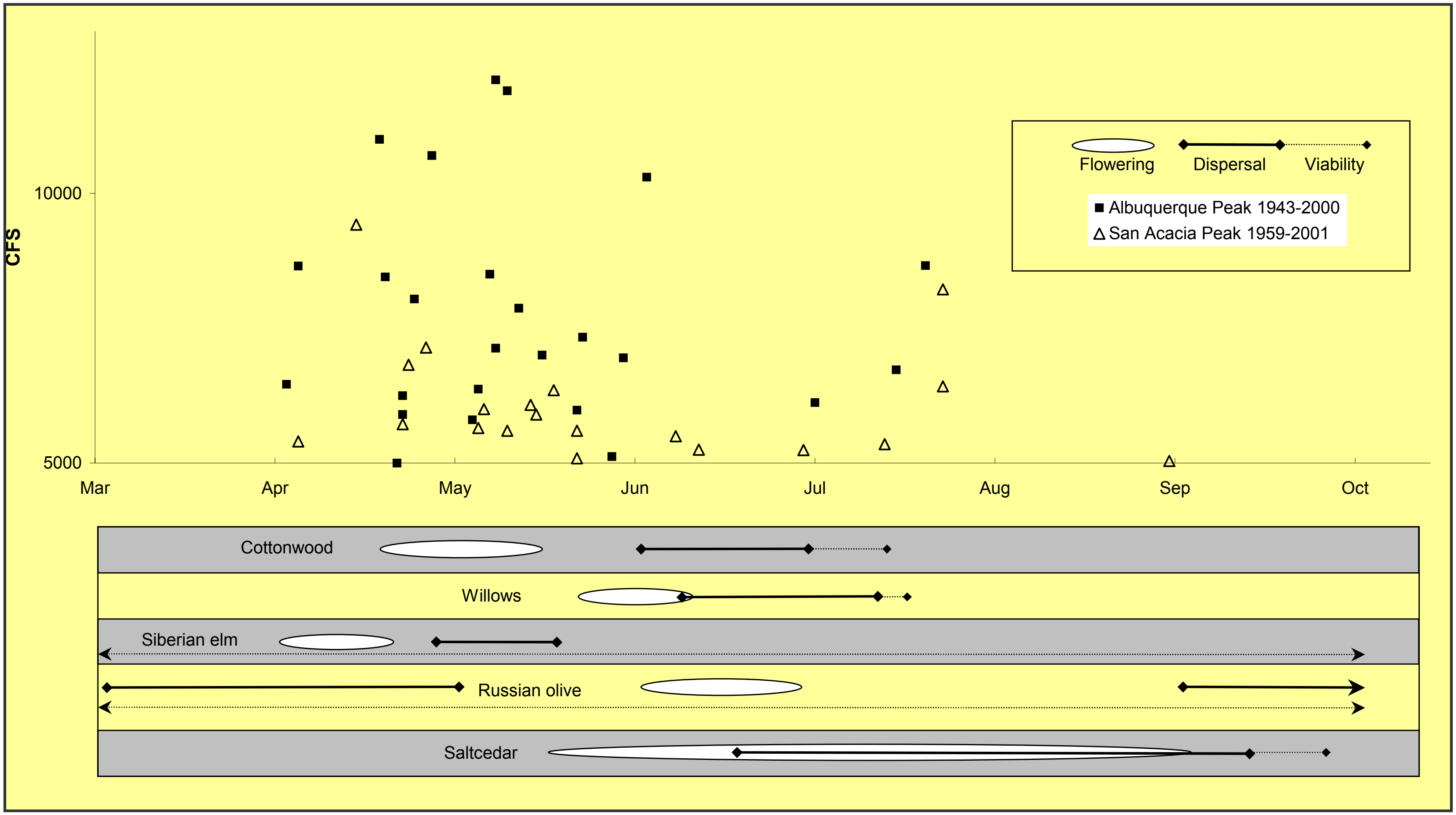

Figure 3-10: Peak annual flow (>5,000 cfs) for gages vs. riparian seed dispersal

3.3.5 Vegetation Trajectory in the MRG Under a Flood Regime

Recent colonization of bars in the Albuquerque area provides an indication of the likely vegetation trajectory for disturbance sites. Vegetation data collected from islands and vegetated bars in the Albuquerque area (Milford et al., 2003) suggests that shrublands with equal amounts of coyote willow and non-natives are most likely to develop (Table 3-4). Communities dominated by cottonwood/willow stands established primarily on the alternate bars and represented about 1 percent of the 541 acres surveyed. Shrubland communities dominated by natives were more prevalent on islands as opposed to the lateral bars. Exotics figure prominently in the overall composition of these communities.

Table 3-4: Percent cover of vegetation groups on bars in the Albuquerque reach

Dominant/Codominant	Cover (%)
Cottonwood/Native	1%
Cottonwood/Mixed	5.2
Cottonwood/Saltcedar	3.2
Non-native	5.2
Coyote Willow/Native	15.8
Coyote Willow/Non-native	10.4
Non-native/Mixed	27.9
Herbaceous	27.2
Bare/Other	4.1

3.3.6 Summary of Overbank Flooding and Vegetation Relationships in the MRG

Cottonwoods and willows evolved to take advantage of the dynamic fluvial and geomorphic patterns that characterize the riparian environment. Specifically, these native plants adapted to episodic flooding that intermittently created sites where new age classes could become established and regenerate the riparian forest community. The periodic overbank flows associated with spring runoff bring the soils to field capacity prior to seed dispersal. The decline of cottonwoods and willows in the Rio Grande bosque could be attributed to the cessation of periodic flood pulses and the introduction of invasive woody exotic species (Howe and Knopf, 1991; Crawford et al., 1993.

A number of techniques have been proposed or discussed that offer options to maintain and/or regenerate cottonwood and willow communities. The techniques include, but are not limited to, hydromodification to provide periodic overbank flooding, mechanical manipulation, and removal of flood flow constrictions that limit reservoir release rates (Sections 4.4 and 4.5). Hydromodification to provide periodic overbank flooding has been proposed for a number of years (FWS, 2003a; Sites Southwest, 2002; Crawford et al., 1999; Ellis et al., 1996). Hydromodification involves either purchasing stored water to provide a designed flow event of a specific rate and duration, or manipulating the natural hydrograph to do the same.

Public health and safety considerations currently limit the release of water from Cochiti and Jemez Canyon Dams such that overbank flooding will not occur in many parts of the middle valley at the maximum release rate (7000 cfs). Overbanking does not occur in many parts of the middle valley (e.g., north of Albuquerque; several miles below the San Acacia diversion dam) because of the relative difference in elevation between the river channel and the riparian floodplain. River flows greater than

10,000 cfs would be required to begin overbank flooding in reaches north of Albuquerque. Such flows would likely threaten levee stability in areas from Bernalillo south to Elephant Butte Reservoir as well as the structural integrity of the San Marcial Railroad Bridge.

Currently, overbank flooding occurs naturally in a limited number of locations without impacting existing infrastructure. Hydromodification, flow manipulation and terrace lowering could be used in some parts of the middle valley to increase overbank flooding and restore riparian habitat (e.g. regenerate cottonwood and willow communities). However, the efficacy and safety of these techniques will require reach specific studies to evaluate the benefits, constraints, and possible unintended consequences to the system. Such studies would determine a range of correctly timed discharges that have the potential to inundate the floodplain. Those sites that have the potential to be flooded and do not present risks to public safety and infrastructure could be prioritized for exotic species removal and grading, if necessary. In the areas where hydromodification could be used to manufacture overbanking events, the impact of such events on selected natural processes should be assessed (e.g., litter decomposition, seed bed preparation, seed germination, plant recruitment, etc) as well as geomorphic aspects of the system (e.g. channel capacity and sediment supply. Overbank flooding in these areas could then offer a mechanism to recruit native plants through natural seedling establishment.

Mechanical manipulation of the floodplain (i.e. terrace lowering) has been proposed as an alternative strategy to replicate disturbances caused by large floods (Auble, 1999; Stromberg, 1999; Sprenger et al., 1999). A number of such projects have been implemented in the middle valley including the Albuquerque overbank project, a portion of the Santa Ana restoration project, and the Los Lunas Project. The results of such efforts in generating and maintaining cottonwood and willow are currently being documented and should be available within the next year or so. Because the cost of moving sediment and disposing of vegetation can be significant, the practicality of implementing these types of projects over large areas may be prohibitive.

Trends indicate that non-native species will continue to thrive in the MRG and certain floods may have the unintended consequence of encouraging their dominance if incorrectly timed. In certain areas of the Rio Grande, revegetation activities that employ active planting techniques to establish native plants are likely to obtain a higher proportion of natives to exotic species.

4.0 RESTORATION TECHNIQUES

Numerous restoration activities have been proposed, planned, and implemented along the MRG. The objectives of these activities vary and include water conveyance efficiency, fish and wildlife habitat improvement, fire hazard reduction, recreation enhancement, ecosystem recovery, water conservation, grazing improvements, and cultural considerations. Some practices serve multiple functions and may directly or indirectly benefit the silvery minnow, flycatcher, and Rio Grande water users. Alternatively, the restoration practices may compete with or be detrimental to either the listed species or water users, based on the type of habitat being restored.

Habitat restoration efforts conducted along large rivers are problematic from a technical, ecological, and sociopolitical perspective (Ziemer, 1999). Most aquatic restoration techniques have been developed for small watercourses or were focused on localized habitat improvement and channel stabilization (NRC, 1992). The efficacy of these techniques for the recovery of pelagic broadcast spawning species has not been demonstrated. Useful techniques for silvery minnow and flycatcher habitat restoration at this large scale should be considered to be in a developmental stage. Moreover, while various habitat restoration efforts have been implemented along the MRG that have demonstrated improvement to aquatic and terrestrial habitat diversity in general, demonstrated benefits to populations of either silvery minnow or flycatcher remain to be proven. Consequently, all techniques presented here should be viewed as experimental. The habitat restoration techniques presented in this document were developed based on field observations, professional judgment regarding the species needs, and restoration projects conducted on other rivers. This list of restoration practices is not meant to be all inclusive and there may be other suitable practices that have not yet been identified.

Because restoration activities in one reach may influence the physical and biological conditions in other reaches, habitat restoration techniques should consider the interactions of hydrologic, geomorphic, and ecological factors from an integrated perspective (e.g., upstream and downstream continuum). The reproductive ecology of the silvery minnow will require coordinated efforts across numerous jurisdictional boundaries, including those associated with landownership and water rights. Habitat restoration techniques that might be applied in the MRG are discussed in Section 4.4 for aquatic environments and in Section 4.5 for riparian vegetation.

4.1 Project-Specific Restoration Considerations

The first step in habitat restoration planning for a listed species is to identify critical limiting factors that prevent their recovery. That is, what specific problem(s) need to be addressed. The biology, ecology, and habitat relationships for the silvery minnow and the flycatcher were discussed in Section 3.0. The section discusses specific high-priority habitat needs that must be met to benefit the conservation and recovery of these two species, defining critical habitat requirements to facilitate the selection of appropriate restoration techniques. The discussion is qualified, recognizing that a level of uncertainty exists with regard to the habitat requirements of these species, especially the silvery minnow. However, sufficient understanding exists to proceed with restoration activities that will likely benefit the species and have the least potential of producing adverse effects.

As the priority habitat restoration needs for the species are determined, the Program will select sites and appropriate techniques to use to address those needs. These decisions must be guided by an array of site-specific considerations and constraints. Subsequent documents, which will be developed by the Subcommittee in the near future, will provide reach-specific guidance on habitat restoration needs and opportunities along the MRG. As more information is gathered regarding specific reaches, plans will be updated, and the direction of the Program activities within that reach could change. The selection of specific restoration projects will be based on benefits to the listed species, feasibility of implementation, water requirements and depletions, sustainability and maintenance, permitting, and 2003 BO are important considerations (Section 5).

4.2 Passive and Active Restoration Approaches

Restoration strategies may be categorized as active or passive. When the factors that cause unsuitable habitat conditions are understood restoration may be achieved by curtailing those activities that are cause degradation (NRC, 2002). This approach is called passive restoration (NRC, 2002). Passive restoration strategies, as applied to river systems, includes techniques that enhance and work with natural channel forming processes, thereby allowing the river "to heal itself" (Gordon et al., 1992). In the absence of human intervention, natural disturbances and ecosystem responses will dictate the speed of recovery for areas undergoing passive restoration (NRC, 1996). For fluvial systems, passive restoration techniques are those that allow channels to regain their natural form and function (Leopold et al., 1992). As used in this Plan, passive strategies utilize the natural processes of the river as they currently exist under the management and climatic conditions. Examples of passive restoration techniques include the removal of lateral confinements that restricts changes in the channel location and the cessation of channel maintenance practices that are counter to the natural tendency of the river processes.

The potential benefits of passive restoration practices can often be augmented by integrating active restoration techniques (see below), such as protecting streambanks to promote the colonization of stabilizing riparian vegetation (Gordon et al., 1992), or in contrast, destabilizing banks to counter the adverse effects of channelization, channel incision, and disconnected floodplains. Because passive restoration focuses on altering, reducing, or eliminating the primary causes or factors that are degrading the system or preventing its recovery, its importance cannot be over emphasized. The National Research Council (2002) suggests that passive restoration is the logical and necessary first step in any restoration program - and in many cases may be all that is required. It is important for the Program to analyze the effects, both positive and negative, that could be realized if passive restoration is implemented. Once this information is gathered and analyzed, more informed decisions can be made regarding priorities for future restoration projects and the techniques that will receive priority support from the Program.

Active restoration practices include engineered approaches to artificially replace some aspect of lost ecosystem structure or function. These are activities that depend more on human intervention and less on natural processes. Most simply, active restoration practices are specific "repair procedures" that are used where it is technically or economically infeasible to allow completely natural processes to address habitat dysfunction (Gordon et al., 1992). Commonly, active restoration practices attempt to restore degraded or dysfunctional systems by combining elements of natural recovery with management activities directed at accelerating the development of self-sustaining and ecologically healthy systems (NRC, 2002). Active strategies, in the context of this Plan, utilize mechanical means to effect a change in the river, assuming that natural processes as they currently exist are unable to effect these changes. Most restoration projects would have components of both active and passive restoration.

Experience with passive and active habitat restoration techniques on the MRG is limited. However, alluvial rivers like the Rio Grande, with deformable beds, high sediment loads, and high width to depth ratios, are sensitive to disturbance and poorly suited to structural modification. Thus, many active restoration techniques may be unsustainable and have a low potential for facilitating recovery (Rosgen, 1996). Not infrequently, such techniques are implemented in opposition to the natural tendencies of a river system; consequently, many tend to fail over the long-term or require periodic maintenance (Gore and Shields, 1995). Nevertheless, active restoration practices can be appropriate to address some short- to mid-term restoration priorities until other appropriate longer-term, low-maintenance alternatives can be implemented. Monitoring the performance and persistence of restoration techniques that are implemented along the MRG will provide a basis for the Program to rank the benefits of these techniques. Passive and active strategies may be combined, and the distinction between these approaches is not always clear. Several potentially useful restoration techniques that may be appropriate for the MRG are discussed in the following sections.

4.3 Channel Maintenance versus Habitat Restoration

Appendix C in the 2003 BA identifies and describes a number of practices that are used to maintain and rehabilitate the river channel (BOR and COE, 2003). These practices are generally identified by the activity conducted in the river. Often, restoration practices are identified in the context of a specific river function they intend to address. For example, high-flow side channels and clearing non-native vegetation with channel expansion are typically implemented to facilitate water conveyance during high-stage flows. Similarly, many river engineering practices (revetments, toe revetment plantings, native material plantings, root wad and boulder placements, windrows, permeable jetties, and curve shaping) function to stabilize banks. Most flood control and river operation activities are designed to protect infrastructure and maintain efficient flow capacity. However, some of these techniques can also be beneficial with regard to habitat restoration, but river and channel maintenance techniques should not be confused with habitat restoration techniques.

4.4 Aquatic Habitat Restoration Techniques

This subsection introduces in-stream and other channel-related restoration practices that can be used primarily to improve silvery minnow habitat. In certain cases, some of these techniques can also benefit flycatcher habitat. As indicated in Section 4.2, projects may be implemented that either work with (passive restoration) or against (active restoration) natural river processes. The adoption of a passive restoration strategy is most likely to provide the best success from a long-term perspective and techniques that promote passive restoration should be considered whenever possible (Section 5). The specific aquatic restoration practices discussed in the following subsections include:

1. Passive Restoration
2. Terrace and bank lowering
3. High-flow, ephemeral side channels
4. High-flow, bank-line embayments
5. Arroyo connectivity
6. Main channel widening
7. Removal of lateral confinements
8. River bar and island enhancement
9. Destabilization of islands and bars
10. Gradient-control structures
11. Woody debris
12. Sediment Management
13. Fish Passage

A particular habitat restoration project may incorporate one or more of these techniques. When designing restoration projects, the performance objectives for the techniques proposed, including benefits for both aquatic and terrestrial resources, need to be defined clearly. These objectives would become the criteria for subsequent monitoring and assessment of the success of the projects. Monitoring the environmental responses to Program-supported restoration projects is essential to help ensure that future projects produce the desired results and avoid unintended consequences.

4.4.1 Passive Restoration

The Rio Grande has a natural tendency to change over time and space. The plants and animals that evolved along the Rio Grande adapted to these dynamic conditions. Drastic changes in form and course occurred during episodic, large floods, and the river has also adjusted its form in response to more normal flooding conditions. Considerable effort was expended in the 20th century to control the flow and course of the Rio Grande. Flow regulation, infrastructure, channelization, regular channel clearing, and emergency flood control measures prevented the Rio Grande from forming the kinds of environments that are associated with dynamic channels. The four primary flow control measures that have been implemented on the Rio Grande include flow regulation, channelization, structural controls (e.g., bridges, diversions, levees), and channel maintenance (island and bar clearing). These features will remain, to some degree, in all reaches; however, alternative measures could be used in some cases that would allow passive restoration to be implemented. Where possible opportunities to allow the river to regain a more natural condition should be explored using the kinds of moderate flows that are expected under the current operational regime.

<u>Purpose of Technique</u>. Adopting a passive restoration strategy would allow the development of natural river features, including bars, islands, side channels, sloughs, multiple thread (braided) channels, and more sinuous channels in reaches where the river has a tendency to form these features (Fig. 4-1). The purpose of promoting these features is to increase the heterogeneity of available aquatic habitats to benefit all life stages of the silvery minnow. In particular, many of these features will provide a greater diversity of flow velocities during high flows than is provided by a straighter channel devoid of obstructions. Furthermore, revegetation of the bars and islands will encourage patches of dense thickets of shrubs and trees of various ages that are attractive habitat for flycatchers.

Considerations. Flow regulation might be used to produce higher-magnitude peak flows to accelerate the natural channel forming process and improve the floodplain habitat, when water is available for this activity. Interagency agreements and preplanning would be necessary to take advantage of these events. However, public safety considerations in Bernalillo, Albuquerque, Los Lunas, and Belen will not permit the large, catastrophic floods of the past. Moderate floods of 5,000 to 7,000 cfs that can be safely passed in those reaches should be optimized to maximize habitat benefits. In contrast, large magnitude floods will inevitably occur in the reaches below the Rio Puerco confluence and substantial changes in the channel and riparian corridor can be expected. Thus, restoration plans for the reaches above and below the Rio Puerco should be developed to accommodate these divergent flow regimes. The reaches above the Rio Puerco are characterized by a more consistent and predictable flow regime than the reaches below the Rio Puerco.

Structural controls fix the location of portions of the channel. For instance, the sections of river adjacent to bridges and diversions must be maintained for public safety and operational considerations. Similarly, development within the historical floodplain has resulted in the need for protective levees, limiting the area over which the river can move. The design limits of the levees must be considered in reaches where passive restoration strategies are adopted, because the increased tortuosity of the channel associated with a braided form results in greater uncertainty in flow trajectory under high-flow conditions. These concerns may be overcome through structural modification of the levees. Levee reinforcement or other structural enhancements may be required to allow passive restoration strategies to proceed. Relocation of levees where useful and appropriate may also be an option.

Habitat Implications. A passive restoration strategy has the potential to provide complex and diverse habitat on a biologically meaningful scale. The aquatic and riparian habitat features that form are sustainable and will contribute to silvery minnow and flycatcher habitat enhancement. The curtailment of channel clearing activities in the Albuquerque reach, coupled with low-flow conditions, has allowed the complexity of the channel to increase. Island formation effectively doubles the length of bankline along the reach of the river where an island develops. Naturally developed islands and sand bars would dissipate stream energy, increase shear stress along banklines, and increase edge complexity. These new edge habitats would have the potential to induce net-zero water velocities and eddies that entrain minnow larvae and eggs and provide additional cover habitat for adult-stage minnows. Indirect benefits to the riparian plant community may arise in association with the increased lateral connectivity and channel movement.

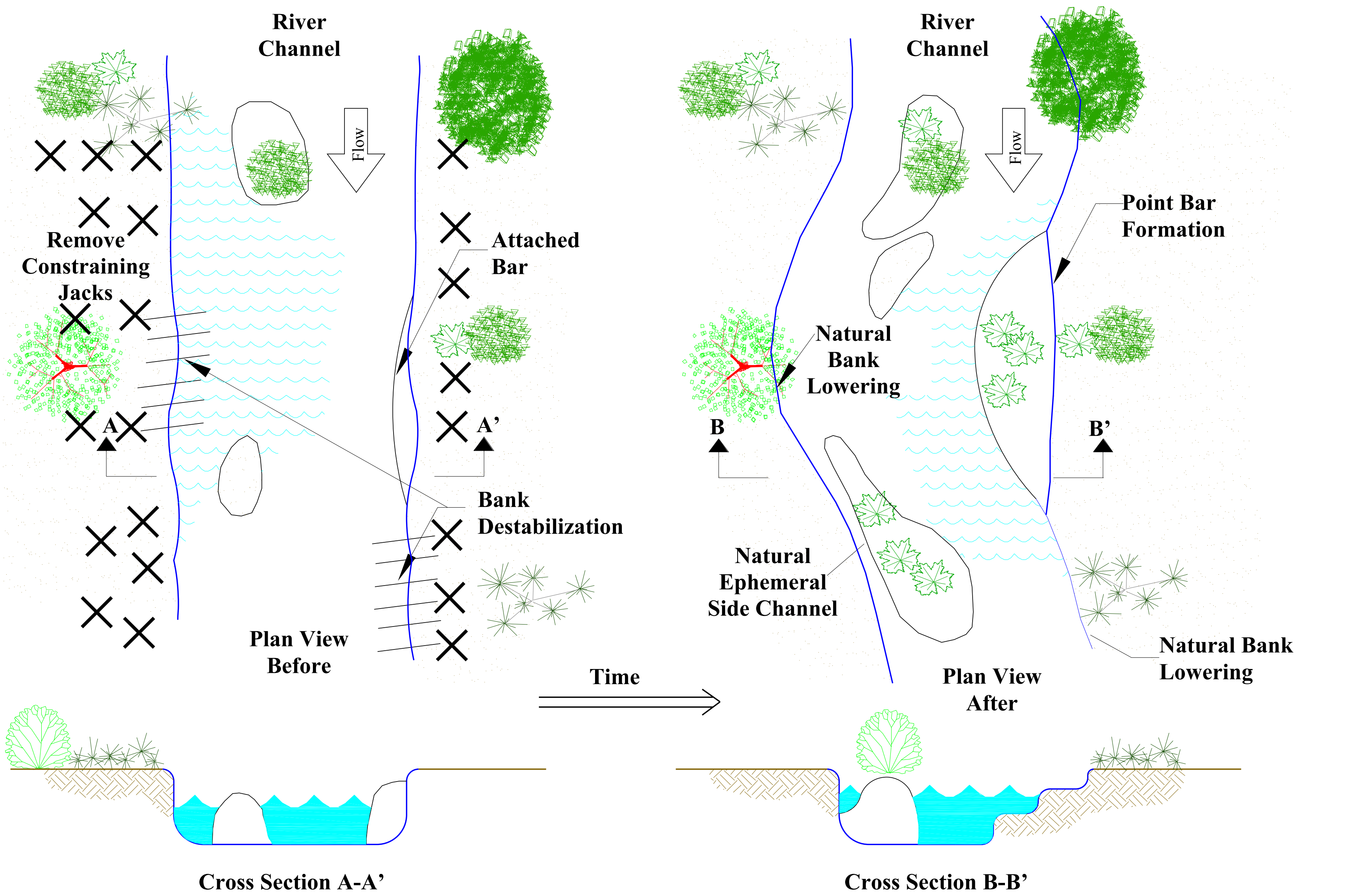

Figure 4-1: Passive restoration, island and bar enhancement

4.4.2 Bank Lowering

Bank (terrace) lowering involves the removal of vegetation and excavation of soils adjacent to the main channel to enhance the potential for overbank flooding. The target elevation for the excavated terrace is below the existing bankfull level (Fig. 4-2). Typically, bank lowering would be appropriate in areas where the channel is incised and/or where banks have aggraded and overbank flooding is limited. The excavated areas are ephemerally inundated and may not flood every year. The Los Lunas Habitat Restoration Project completed in 2003 included bank lowering, excavation of several embayments (see Section 4.4.4), and a side channel (see section 4.4.3). Figure 4-3 is an aerial view of the Los Lunas project area.

Purpose of Technique. Historically, portions of the Rio Grande floodplain were regularly inundated during high flows in the spring, and less frequently during summer storm flows. Infrequently, large flood events resulted in changes in the location of the channel and broader-scale inundation. Bank and terrace lowering is intended to increase the frequency and extent of overbank flooding at selected sites. In addition, bank excavations are intended to replace the processes once performed by large floods that removed vegetation and deposited fresh or bare sediments. Extensive and long-duration flooding may have provided ephemeral nursery habitat for silvery minnow.

Considerations. The configuration (elevation and plan perspective) of the overbank area should be determined by the performance objectives for the project (e.g., riparian or aquatic emphasis). For example, bank lowering at a given location could be designed to foster native plant growth and/or provide shallow aquatic habitat. Design considerations for bank lowering projects to benefit flycatcher habitat are mostly related to establishment and maintenance of appropriate vegetation. The implications of bank lowering and subsequent increases in the frequency of flooding for vegetation recruitment and plant community development are discussed in Section 3.3.4.

Bank lowering projects designed to increase retention of silvery minnow eggs and larvae must consider the volumes of water to be moved into the overbank area, retention times, and the potential for egress of larvae or juveniles to the main channel. In this case, a proportionate amount of water would need to flow into the lowered overbank area and intercept a biologically significant quantity of eggs. Eggs and larvae must then be retained for a sufficient duration of time. Because the magnitudes of spring flow peaks that induce spawning vary annually, design and construction of overbank areas should accommodate a wide range of flows likely to induce spawning. Another strategy may be to lower the banks at multiple sites to accommodate different flows. The residence time of the eggs and larvae on the floodplain will depend on the net velocity of the water, which is affected by the depth, channel roughness, and gradient of the inundated area and the length of the facility. The overbank area must also ensure that an outlet to the main channel persists during periods of declining flood flow volumes (i.e., declining river stage). Overbank areas that lack an adequate outlet design would result in stranding the eggs and larvae on the floodplain. Thus, the overbanks areas must be longitudinally extensive along the river or occur at high densities (i.e., multiple or larger sites).

For silvery minnow habitat restoration projects, construction tolerances for the bank elevations are likely to be narrow to achieve the target flow velocities, and sedimentation will reduce their effectiveness. Design of these facilities is complicated by the difficulties in predicting the impact of vegetation colonization on sediment accumulation rates. Because sedimentation will affect the longevity of the overbank areas, the sediment dynamics for the reach associated with other upstream and downstream restoration projects should be considered. The eventual formation of natural levees at the channel-bank interface may prevent egress. Moreover, as sediment accretes in the overbank areas, flood frequencies will be reduced. Project designs should include expected lifespan and/or maintenance schedules. These issues are of less concern for projects aimed at flycatcher habitat improvement. Vegetation and sediment removal required to maintain silvery minnow habitat function may negatively impact flycatcher habitat.

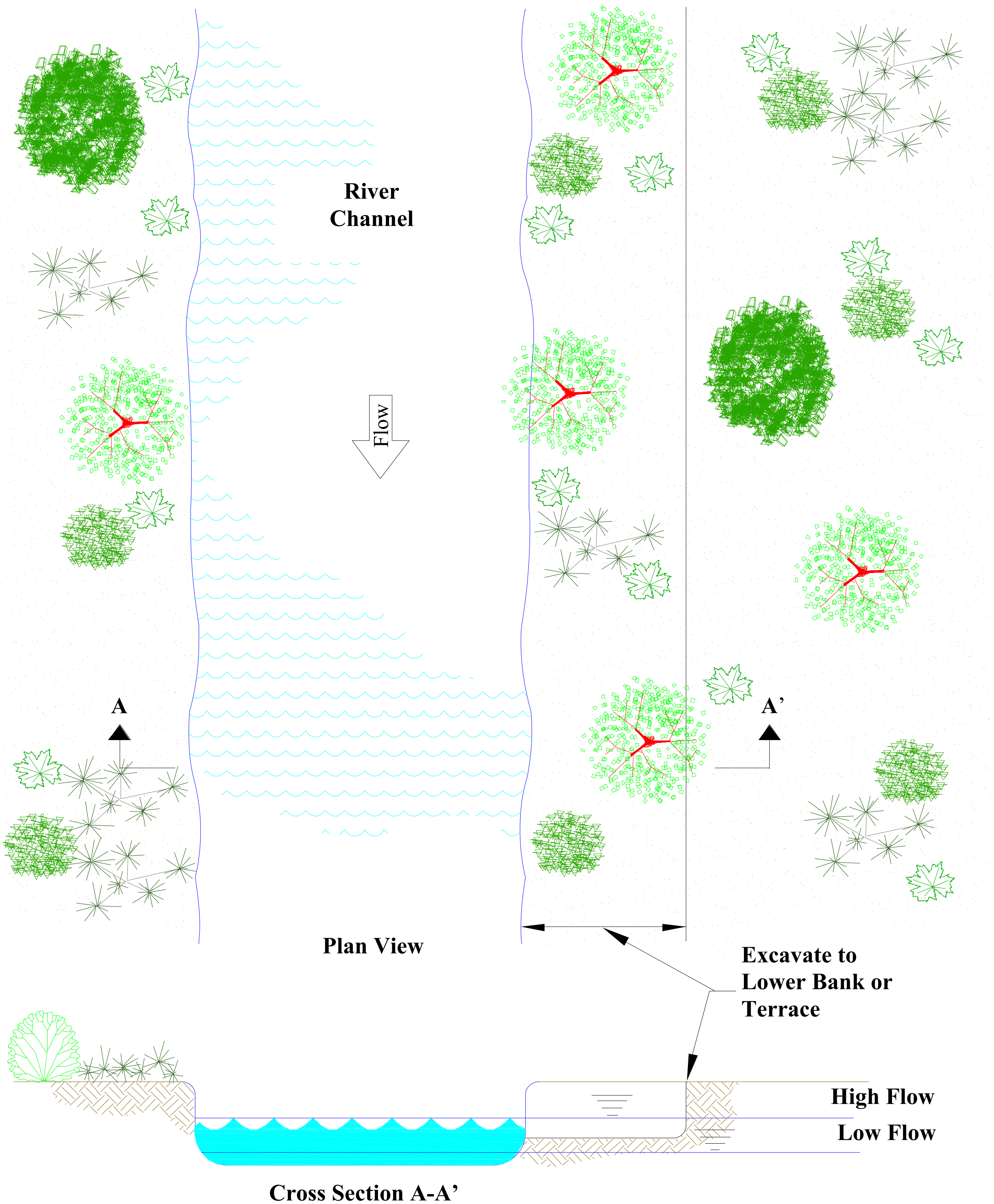

Figure 4-2: Bank Lowering

Figure 4-3: Bank lowering and bank-line embayments: Los Lunas Habitat Restoration Project
(Photo courtesy of Reclamation)

Existing vegetation, the presence of jetty-jacks, and the excavation volumes should be considered in site-selection. Vegetation clearing methods are discussed in Section 4.5.1. The disposition of the excavated spoils should be considered, including the potential positive and negative impacts of adding excavated sediments to sediment-deficient river reaches.

Habitat Implications. Bank lowering has been proposed as a means to enhance silvery minnow and/or flycatcher habitat. Because the overbank areas would not remain flooded for significant durations, they are not expected to provide aquatic habitat for adult silvery minnows. However, if properly maintained and of sufficient scale, the overbank areas could improve retention of eggs and larvae. Depending on the character of the existing vegetation, flycatcher habitat may be increased through the establishment of potentially more desirable vegetation. Indirect benefits to the downstream aquatic community such as the mobilization of organic materials may arise from the increased lateral continuity between the river and floodplain.

4.4.3 Ephemeral Side Channels

Ephemeral side channels are low-gradient, flow-through channels that are connected to the main (perennial) river channel. Side channels are intended to carry flow from the main channel at designated discharges, typically during high-flow events such as spring runoff (Fig. 4-4). During these events, the side channel would carry water at a lower velocity than the main channel and may include ponded areas with near-zero flows (Figure 4-5). Design variations may also include wetlands and ponds, connections to abandoned river channels and oxbows, and embayments or sloughs that have one-way connections to the main channel. Side channels differ from bank lowering in that water is discretely channeled and not spread out on a lowered floodplain.

Purpose of Technique. As with bank lowering, this technique is intended to replicate historical channel conditions along the Rio Grande associated with overbank flooding and avulsion channels in a braided system. The high flow events resulted in the creation of shallow, ephemeral, low-velocity aquatic habitats that were probably used by the silvery minnow, at least during some life stages. Depending on the duration of flow in the side channel, the local elevation of the water table may increase, which could result in increased density of vegetation along the channel.

Considerations. Construction of a side channel will require removal of existing vegetation and excavation of bank materials. The aggradation and degradation trends of the main channel should be considered when developing side channel projects. The lower-velocity flows in the side channels will result in sedimentation and potential plugging of the side channel. The effective biological influence of side channels on the silvery minnow depends on the balance between the low velocities, slope, and sediment deposition. Thus, these structures may have limited lifespans or require periodic maintenance.

Depending on the design, bank stabilization may be required to maintain the inlet, and projects should be constructed along reaches where the adjacent channel is projected to remain relatively stable. If the primary objective of a side channel facility is to slow the transport of silvery minnow eggs and larvae, consideration should be given to the amount of water diverted, retention times, and the potential for egress by the young minnows. Thus, a detailed analysis of the length/velocity relationships at various flows may be required. Because of the annual variations in spring runoff, side channels should be designed to accommodate a wide range of flows to encourage recruitment every year. As with bank lowering, multiple side channels could be constructed to accommodate a wide range of flows.

Other items to consider in side channel design include techniques for maintaining open channels, increasing the complexity of channel edges with snags/woody debris and as a food substrate, and planting willows for flycatcher habitat and aquatic cover/shade. The topography, existing vegetation, and occurrence of jetty-jacks should be considered in site selection. Vegetation clearing methods are discussed in Section 4.5.1. Provisions for disposal of the spoil will be required to accommodate excavation of the new channel.

Habitat Implications. Side channels have been proposed primarily to enhance silvery minnow habitat, but they could also promote riparian vegetation regeneration potentially beneficial to the flycatcher. Specifically, if the channels are long enough and provide sufficient residence times for hatching and larval development, they could increase retention of young silvery minnows in the project area. Most side channels will dry during low flows and will not provide aquatic habitat for adult silvery minnows. Depending on the character of the existing vegetation, flycatcher habitat may be enhanced if vegetation density along side channels increases.

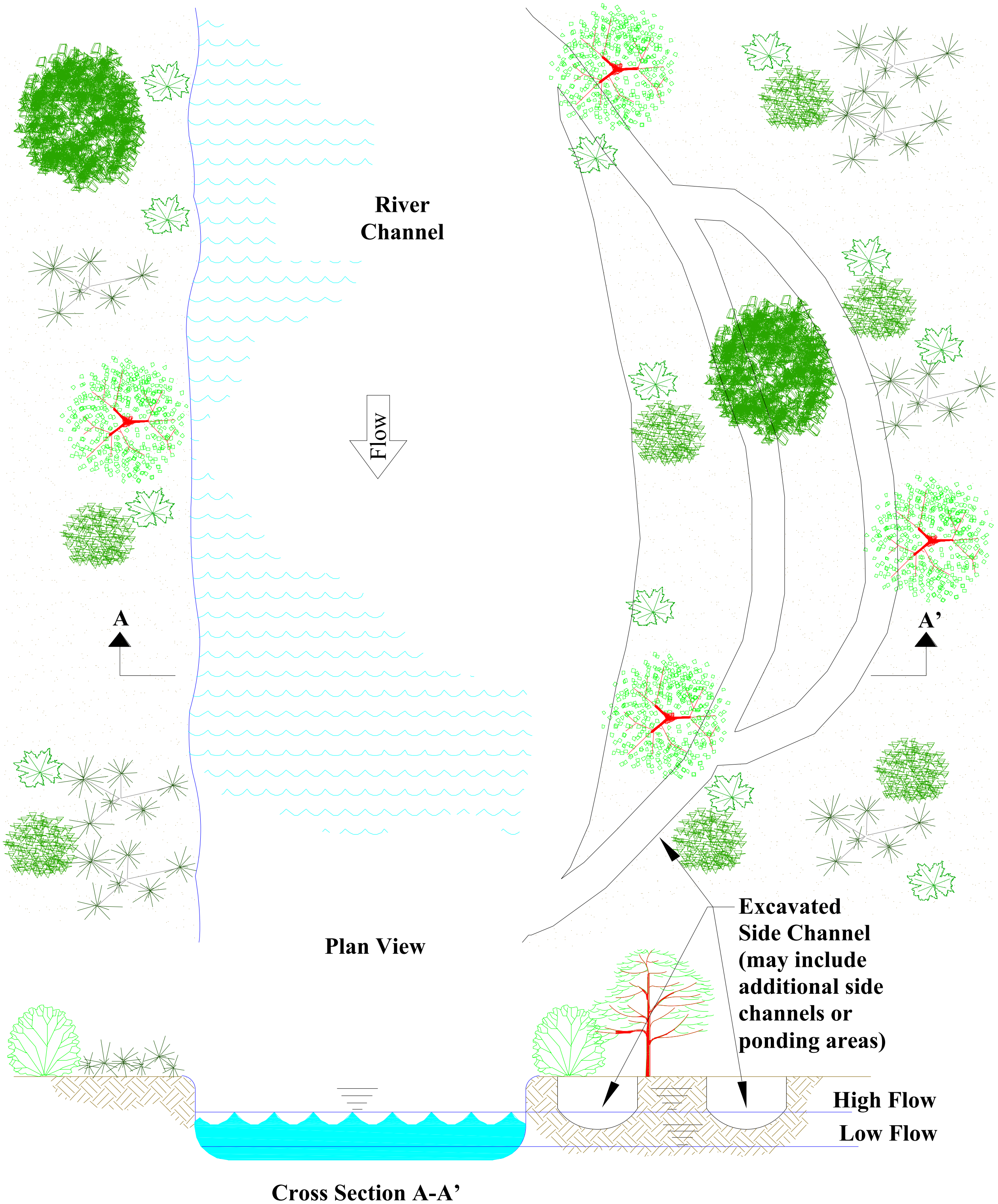

Figure 4-4: Ephemeral side channels

Figure 4-5: Natural ephemeral side channel during spring flow
(Photo courtesy of M. Marcus, Tetra Tech)

4.4.4 Bank-line Embayments

Bank-line embayments, as presently conceptualized, are areas cut into the banks where water from the main channel enters primarily during higher-flow discharge events such as floods (Fig. 4-6). The targeted discharge volumes and velocities are associated with events that produce spawning by silvery minnows (Porter and Massong, 2003). Embayments differ from high-flow ephemeral side channels (Section 4.4.2) in that they exchange water with the main channel across a broad section of bankline, rather than at discrete inlets and outlets.

Purpose of Technique. Bank-line embayments are primarily intended to retain drifting silvery minnow eggs and fry, retarding their downstream displacement during spawning (Porter and Massong, 2003). Secondarily, embayments are intended to provide rearing habitat and to enhance food supplies for developing silvery minnows (Porter and Massong, 2003).

Considerations. Embayments are typically configured to slope upward away from the main channel, allowing them to function over a range of discharge volumes. While the design, development, and sustainability of these structures have not been definitively tested, available information suggests that the more effective designs have inlet width of 30 to 50 meters, with embayment lengths of 1.5 to 2.0 times the width of the inlet mouth (Porter and Massong, 2003). Other recent preliminary relationships reported by this study of embayments along the MRG (Porter and Massong, 2003) indicate the following:

1. Large drift zones were more effective in egg retention.
2. The depth, shape, location, and angle of the embayment inlet influence the intake and discharge of water and determine the effectiveness of egg entrainment and retention.
3. Inlets fill with sediment, which affects the longevity and quality of embayments as restoration sites.
4. Embayment structures retain considerable organic debris that could serve as potential food for both young and adult silvery minnows.

The shallow, quiet waters associated with bank-line embayments can also provide favorable habitat for potential competitors and predators of the silvery minnow (M. Porter, BOR, personal communication). Deep backwaters have been found to become attractive nuisances, providing habitat for fish species, that have the potential of preying on silvery minnows, including the red shiner, channel catfish, white sucker, and yellow perch. These embayments should be designed to create suitable egg retention habitat during spring runoff, with only minimum water intrusion during base flows.

Habitat Implications. The lack of in-stream structures that would produce complex eddy currents, backwater areas, and features that generate near-zero flows can result in excessive downstream drift of silvery minnow eggs during spawning events (Platania and Altenbach, 1998). Preliminary data from the Rio Grande indicate that bank-line embayments may retard the downstream drift of artificial eggs and silvery minnow eggs (Porter and Massong, 2003). Thus, bank-line embayments provide a potential restoration approach with regard to lessening downstream displacement impacts. The frequency and spacing of embayments that would be required to have a measurable effect on the silvery minnow population is unknown. Additional research is needed to understand the implications with regard to predatory interactions.

In the absence of maintenance, embayments will fill with sediment and be colonized by vegetation that is expected to provide flycatcher habitat, as the areas become wetlands.

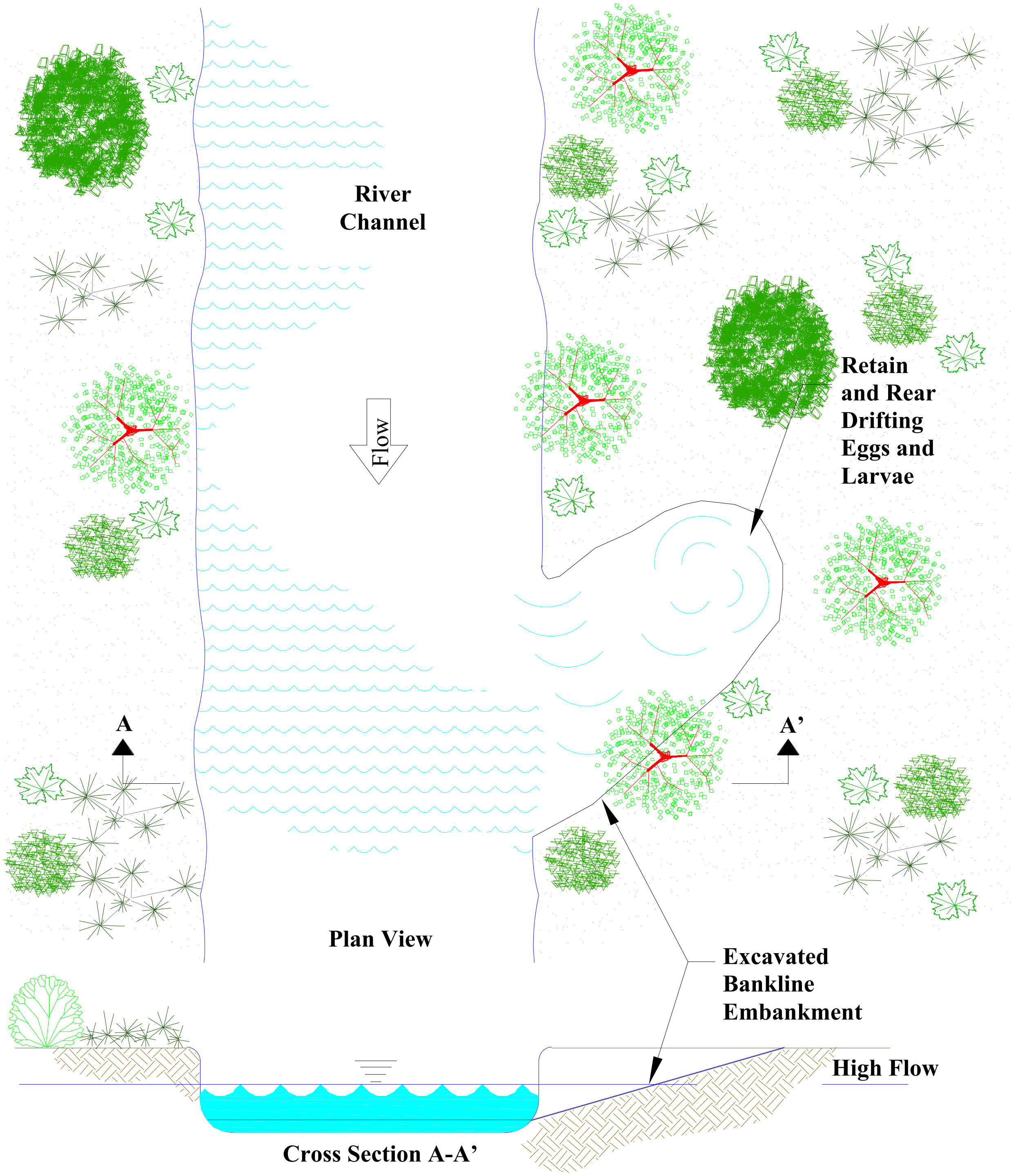

Figure 4-6: Bank-line embayment

4.4.5 Arroyo Channel Reconnection

Sediment plugs have formed at the mouths of some of the tributary arroyos to the Rio Grande, resulting in stranded arroyo channel beds that are not at grade with the Rio Grande channel. Vegetation has invaded the arroyo channels restricting water and sediment delivery to the Rio Grande. In other instances, arroyos are blocked from the river by levees, irrigation channels, and the LFCC. The arroyo channel reconnection technique involves the clearing of vegetation and/or excavation of a pilot channel to bring the stranded arroyo to grade with the main channel of the Rio Grande.

Purpose of Technique. Biological surveys suggest that silvery minnow eggs and larvae aggregate in eddies associated with the mouths of arroyos (M. Porter, BOR, personal communication). Reconnecting arroyos to the Rio Grande is viewed as a potential opportunity for increasing egg retention sites, which could retard the downstream drift of developing silvery minnows. In addition, arroyo channel reconnection may locally increase water and sediment supply to the Rio Grande. Sediment accumulations in the main channel that originate from arroyos can act as local grade controls (LaGasse, 1981).

Considerations. The causative factors behind the stranding of the arroyos and sediment dynamics in the Rio Grande are not well understood. Natural cycles of arroyo cutting and filling suggest that many of the disconnected arroyos may eventually reconnect to the river in response to complex climatic and watershed interactions. For instance, an arroyo watershed may not have experienced a threshold storm capable of mobilizing sediment in the recent past. In other cases, sediment control dams on the arroyo may affect peak flow and sediment dynamics at the channel confluence. Thus, the sustainability of this technique is difficult to predict without a clear understanding of the factors controlling sediment dynamics in an individual arroyo.

Reconnecting the arroyo channel may have downstream impacts with regard to the potential for the addition of sediment to the river, especially if headcutting is initiated in the arroyo. Cutting a pilot channel may also accelerate upland erosion as the system adjusts to a new base level. Accelerated downcutting in the arroyo could result in the formation of a local fan or bars in the Rio Grande channel. The implications of the formation of fans should be considered relative to the safe passage of flood flows. As with the other techniques that involve vegetation removal and excavation, the potential for habitat destruction exists and provisions for disposal of spoils must be considered. Vegetation removal from arroyos should consider the type of vegetation and the implications for protecting adjacent stands from fire. The upstream impacts of arroyo headcutting should be considered with respect to watershed land management issues.

Habitat Implications. Arroyo reconnection could lead to localized increases in the number of egg retention areas that could retard the downstream drift. Reconnection of arroyos increases the supply of sediment to the river and alluvial fans can act as local grade controls. The addition of sediment to the river may influence the development of islands and bars in the Rio Grande, which could potentially increase aquatic habitat diversity.

4.4.6 Channel Widening

Channel widening involves excavation of the banks and lateral expansion of the active channel area (Fig. 4-7). This technique is differentiated from bank lowering by excavating deep enough to maintain a wetted perimeter under conditions of less-than-bankfull discharges. When implemented for the purpose of channel maintenance, channel widening will produce engineering benefits, including conveyance efficiency and reductions in overbank flooding. The channel widening technique discussed here should not be confused with the channel rectification and maintenance activities.

Purpose of Technique. Channel widening is intended to reduce average flow velocities in reaches where low-velocity aquatic habitat potentially beneficial to silvery minnows is lacking throughout most of the year. Moreover, channel widening could allow the Rio Grande to develop more diverse channel and floodplain features if islands and bars form. Channels that have narrowed significantly and where banks are stabilized by vegetation may develop a greater variety of aquatic habitats following channel widening.

Considerations. Depending on project objectives and site-specific geomorphic trends, channel widening projects may require continued maintenance to keep the channel from narrowing. Thus, the cause(s) of channel narrowing must be clearly understood to ensure that the project is sustainable or to make provisions for maintenance. Management of the sediment from the excavation is an important consideration. In areas where the channel has narrowed because of too much sediment, in-channel disposal is probably a poor option.

Channel widening could affect downstream flooding, and this will likely require a detailed analysis of public safety considerations. Channel widening projects could include the removal of jetty jacks and vegetation and be designed to leave the channel with variable depths to initiate the formation of bars and islands.

Habitat Implications. The actual beneficial effects of channel widening on silvery minnow habitat will require detailed geomorphic analysis and monitoring. Channel widening may increase the total area of lower-velocity, shallow habitat for young-of-year and adult silvery minnows, but it is uncertain whether velocities during spring spawning flows will reduce downstream transport of eggs and larvae. Section 3.1.4.2 discusses flow velocities relative to the habitat requirements of the silvery minnow at various life stages. The potential for channel widening to create more diverse channel conditions ultimately rests in designs that result in the addition of net-zero and low-velocity habitats.

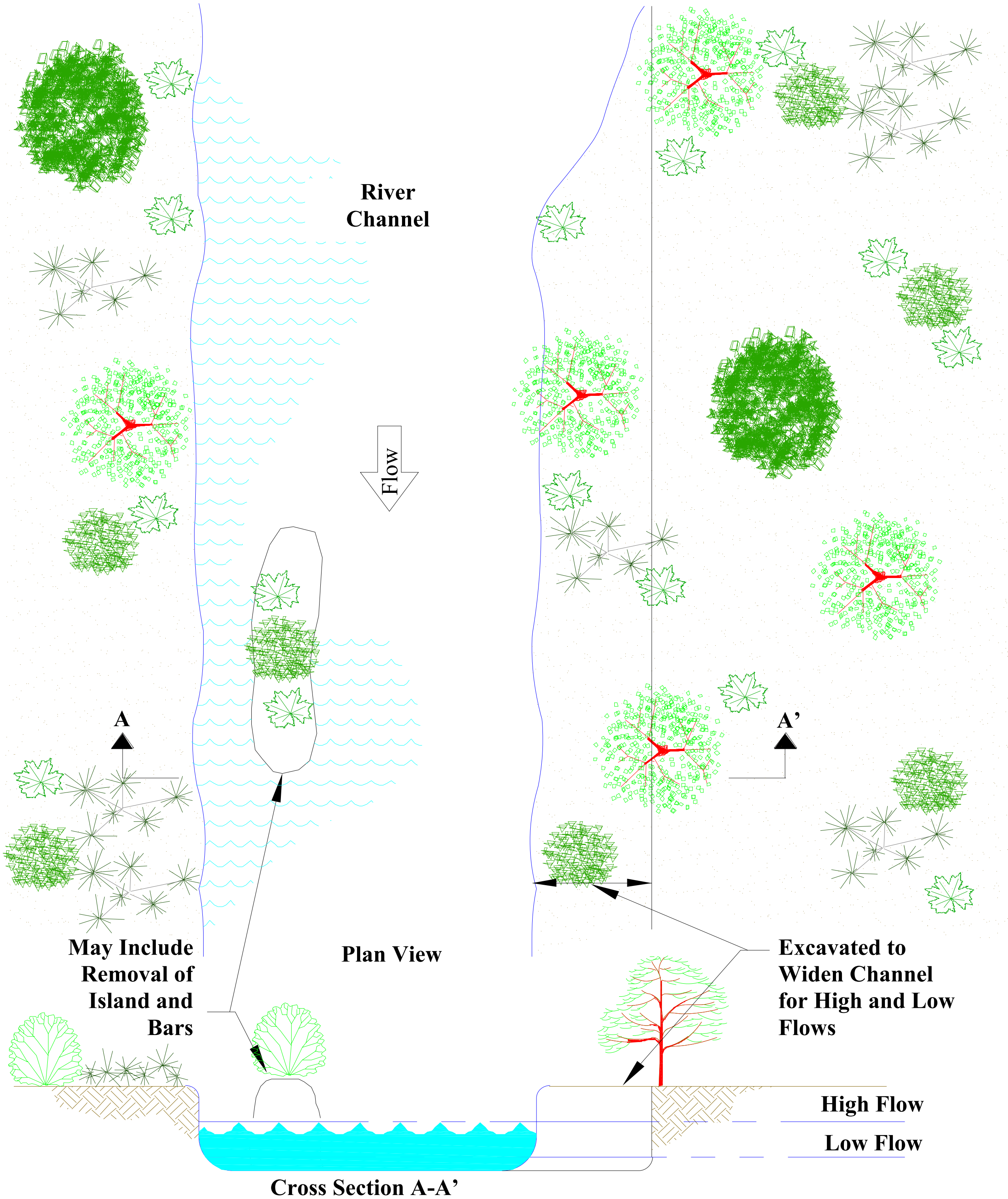

Figure 4-7: Main channel widening; island bar removal

4.4.7 Removal of Lateral Confinements

Removal of lateral confinements includes activities that reduce or eliminate structural features and maintenance practices that decrease the potential for the channel to erode its banks (Fig. 4-8). This practice would be implemented as part of an overall passive restoration strategy. Features that confine the Rio Grande channel include, but are not limited to, (1) jetty jacks that were originally placed to rectify (straighten and narrow) the floodway; (2) artificially straightened or rectified portions of the channel that are being maintained; and (3) woody vegetation that has stabilized river banks under the current flow regime and channel configuration. On a larger scale, bridges, diversions, and hard-engineered features (e.g., grade control structures) are point controls that fix the location of the channel, and some flood control levees may also be considered as lateral confinements. However, this restoration technique is not intended to address these hard-engineered features.

Purpose of Technique. The purpose for removing lateral confinements is to allow the Rio Grande to develop more diverse channel and floodplain features. This technique is proposed on the assumption that floodway rectification and subsequent maintenance have prevented diverse channel features from forming, thereby reducing habitat diversity. Thus, reaches that are not laterally confined will ultimately develop a greater variety of sustainable aquatic and riparian habitats.

Considerations. Because the technique of removing lateral confinements ultimately relies on the river to do much of the work, the flow regime following this action will determine the rate of change in the channel and floodplain. Nonetheless, project costs can be reduced where restoration benefits are not urgent by using passive techniques and letting the river energy do the work. In many areas, the banks are stabilized by vegetation and its removal may be required. Bank excavation and river training practices may be required to initiate destabilization of the banks. Such actions may increase local sediment loads resulting in transient changes in sediment dynamics. In all cases where removal of lateral confinements is considered, detailed evaluations regarding public safety concerns will be required.

Habitat Implications. The removal of lateral confinements should allow the river to assume a more natural form. The direct benefits of this restoration practice may not be immediate, but potentially large scale and sustainable habitat restoration can be achieved. Assessing the benefits to the silvery minnow habitat will require detailed geomorphic analysis. Nonetheless, the creation of more diverse channel conditions should ultimately result in additional net-zero and low-velocity habitats for the silvery minnow. Moreover, flycatcher habitat could improve as point bars are colonized by dense riparian vegetation.

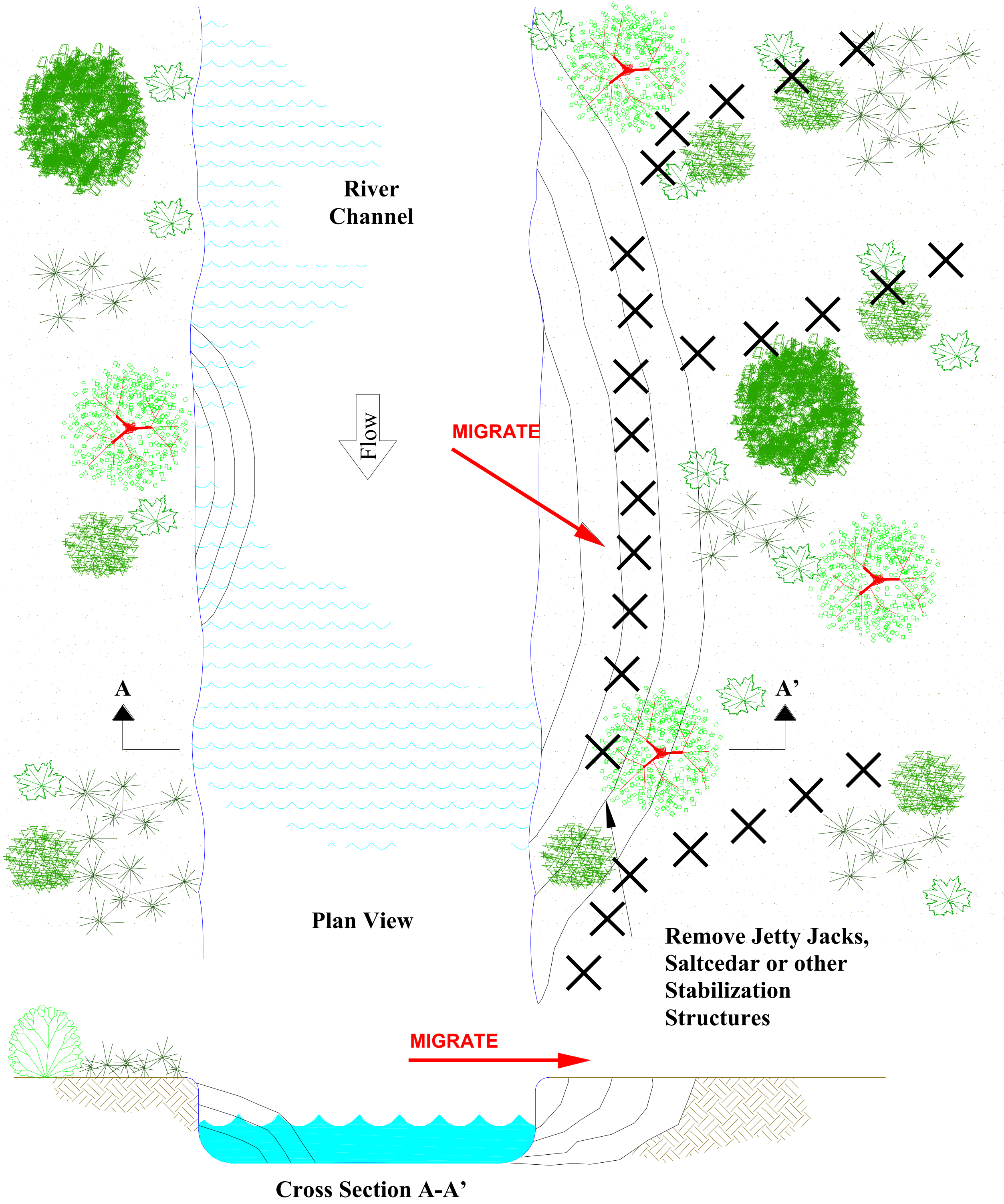

Figure 4-8: Removal of lateral confinements

4.4.8 River Bar and Island Enhancement

Bars and islands are common features in rivers, especially those with abundant sediment supply and erratic flow regimes. Bars are transient, unvegetated, sediment deposits in the channel that are submerged at bankfull stage, while islands are vegetated medial bars that are relatively long-lived features (Brice, 1964; Bridge, 1993). Point and alternate bars may vegetate, causing the channel to narrow. Islands form readily in the Rio Grande during extended periods of relatively low flow. A substantial number of islands and vegetated bars have formed during the low flows that have characterized the late 1990s. Bar and island enhancement is part of a passive restoration strategy for reaches where these features form (Fig. 4-1). The technique involves the elimination of channel maintenance and making adequate provisions for allowing the islands and bars to form. Conceivably, river bars and islands could be developed with different levels of intervention, and small-scale manipulations might include installation of snags or other types of structures (vanes, weirs or deflectors) in the channel to initiate bar formation and eventual island development.

Islands may expand or contract in response to changes in flow regime and sediment supply. Aggradation, and growth, occurs during high-flow sedimentation promoted by the increased roughness associated with vegetation. Islands are semi-permanent and may be removed depending on the magnitude of the flow and developmental stage of the vegetation. Once established, bars and islands are self-promoting as flows diverge and converge, alternately aggrading, scouring, and depositing sediments downstream to form new bars (Bristow and Best, 1993). In meandering and straight channels, point and alternate bars may become vegetated and restrict the channel width, thus encouraging erosion on the opposite bank during high-magnitude flow events.

Islands may be seasonally inundated and function in a manner similar to floodplains. Riverside levees form on the margins with lower lying basins and high-stage channels in the interior. High-flow events may breach the island levees and reoccupy the interior channels. On the downstream ends of the islands, interior channels may form sloughs and backwaters depending on the flow conditions.

Purpose of Technique. In many areas of the MRG, the Rio Grande has a tendency to form islands and vegetated bars (BOR, 1975). River bars and islands increase the variety of aquatic habitats by creating a more complex channel configuration, including backwaters, shear zones adjacent to the bar/island banks, and convergent and divergent eddy zones at the upstream and downstream ends of the islands. This heterogeneity of aquatic habitat is likely to benefit all life stages of the silvery minnow. Furthermore, the colonization of bars and islands results in the development of stands of vegetation of various ages that may have attributes of flycatcher habitat.

Considerations. In the past, considerable effort was expended to clear bars, islands, and debris from the channel. These activities were conducted on a regular basis between about 1950 and the mid 1980s by BOR and, to a lesser extent, the MRGCD to maintain the floodway through the channelized reaches (BOR 1975, 1993). From an operational perspective, the occurrence of islands is problematic because they reduce channel capacity and may cause increased erosion on the opposing banks during high flows. If the opposing bank is fixed and the bed is deformable, the channel may incise. Enhancement of bars and islands may not be appropriate for certain reaches and may not be practical due to existing channel constraints. In other reaches, this may be an appropriate technique to increase overall aquatic and riparian diversity, particularly when used in conjunction with other techniques such as removal of lateral constraints and channel widening.

Establishment of stable, vegetated bars and islands may indicate a change in river morphology trending from a single thread to a braided channel (Knighton and Nanson, 1993). Initial instability may lead to downcutting as the channel is constrained by the islands and bars and resistant banks. However, this degradation will facilitate undercutting of the outer banks and allow the channel to make lateral adjustments. Lateral movement of the channel could jeopardize levees and increase the potential for flooding. Thus, public safety concerns must be carefully evaluated through geomorphic and hydraulic analyses.

River bar and island projects may include jetty jack removal, levee setback and relocation or reinforcement of the existing spoil bank levees. Measures to ensure native plant success need to be included in the enhancement plans for islands because non-native vegetation could also become established. Exotic vegetation clearing and revegetation methods are discussed in Section 4.5.

Habitat Implications. Islands and bars may enhance silvery minnow habitat and potentially promote the recruitment of riparian vegetation that could benefit the flycatcher. Islands effectively double the length of bankline along the reach of the river where they occur. Islands and sand bars dissipate stream energy, increase shear stress along banklines, increase edge complexity, and provide active floodplain functions. These edge habitats have the potential to induce net-zero water velocities and eddies that entrain minnow larvae and eggs and provide additional cover habitat for adult minnows. Flow environments in the aquatic microhabitats associated with bars and islands have not been fully characterized: therefore, the total amount of bankline necessary to capture a biologically significant quantity of eggs remains undefined.

Indirect benefits to the riparian plant community may arise in association with the increased lateral connectivity and channel movement.

4.4.9 Destabilization of Islands and Bars

The Rio Grande has a natural tendency to form islands and bars in some reaches (Section 2.5). The shape, number, and locations of bars is related to the nature and supply of sediment and flow regime (Bristow and Best, 1993). Bars are either transient features that are formed on the receding limb of a flood and later removed during large flows or more persistent features if prolonged low flow conditions persist after their initial formation. The establishment of vegetation increases the likelihood that bars will persist. Once stabilized the bars reduce the capacity of the channel if the banks are resistant to erosion. Where no lateral constraints exist, the width to depth ratio of braided channels (with bars) is greater than the width to depth ratio of single thread channels. Thus, unconstrained braided channels should result in lower overall flow velocities and increased tortuosity compared to single thread channels. However, where the banks constrain the channel the formation and stabilization of bars can result in channel degradation and increased velocity.

Bars and islands can be destabilized by physical disturbance (discing, mowing, root-plowing, and raking) to remove vegetation; destabilizing these features makes them mobile during high flows (Fig. 4-9). The 2003 BO indicates that "action agencies, in coordination with parties to the consultation and in consultation with the Service, shall prevent encroachment of saltcedar on the existing channel and destabilize islands, point bars, banks, or sand bars in the Angostura, Isleta, and San Acacia Reaches."

Purpose of Technique. This technique may be implemented to address channel narrowing and degradation produced when flows are constricted by in-channel structures, including islands and bars. Destabilization would then allow river processes to produce a wider channel through a natural, scouring process (FWS, 2003). The FWS (2003) defines the purpose of island, bar, and bank destabilization, as well as other reasonable and prudent alternatives (e.g., relocation of the San Marcial railroad bridge, overbank flooding, and increased discharge and transport of sediment into the MRG), as intending to maintain or improve the quality and quantity of habitat available for the silvery minnow and flycatcher.

Considerations. The 2003 BO stipulates that the methods used and areas proposed for island, bar, and bank destabilization should be agreed upon by the Service, Reclamation, the Corps, appropriate pueblos, and affected private landowners. They also advise that this activity should not adversely affect flycatcher habitat and that these actions should be undertaken when reaches are dry.

When designing island, bar, and bank destabilization projects, additional consideration should be given to potential long-term maintenance that may be needed to maintain the project. Implementation of this technique should avoid adverse modification of silvery minnow critical habitat by ensuring that primary constituent elements are provided or restored.

Habitat Implications. Island, bar, and bank destabilization may help alleviate adverse modifications to silvery minnow critical habitat by providing for the necessary habitat components of Primary Constituent Elements 1 and 2 defined in the 2003 BO. The first of these two elements includes the need for actions that form and maintain backwaters, shallow side channels, pools, eddies, and runs of varying depth and velocity (FWS 2003). The second element advocates the presence of eddies created by debris piles, pools, or backwaters, or other refuge habitat within unimpounded stretches of flowing water of sufficient length (i.e., river miles) to provide variations in habitat [and] with a wide range of depths and velocities of water in the river. In some reaches, island removal would contribute to reducing average channel depth and widening the channel. In general, island and bar destabilization and removal projects should result in the creation of more complex, diverse habitat (FWS, 2003).

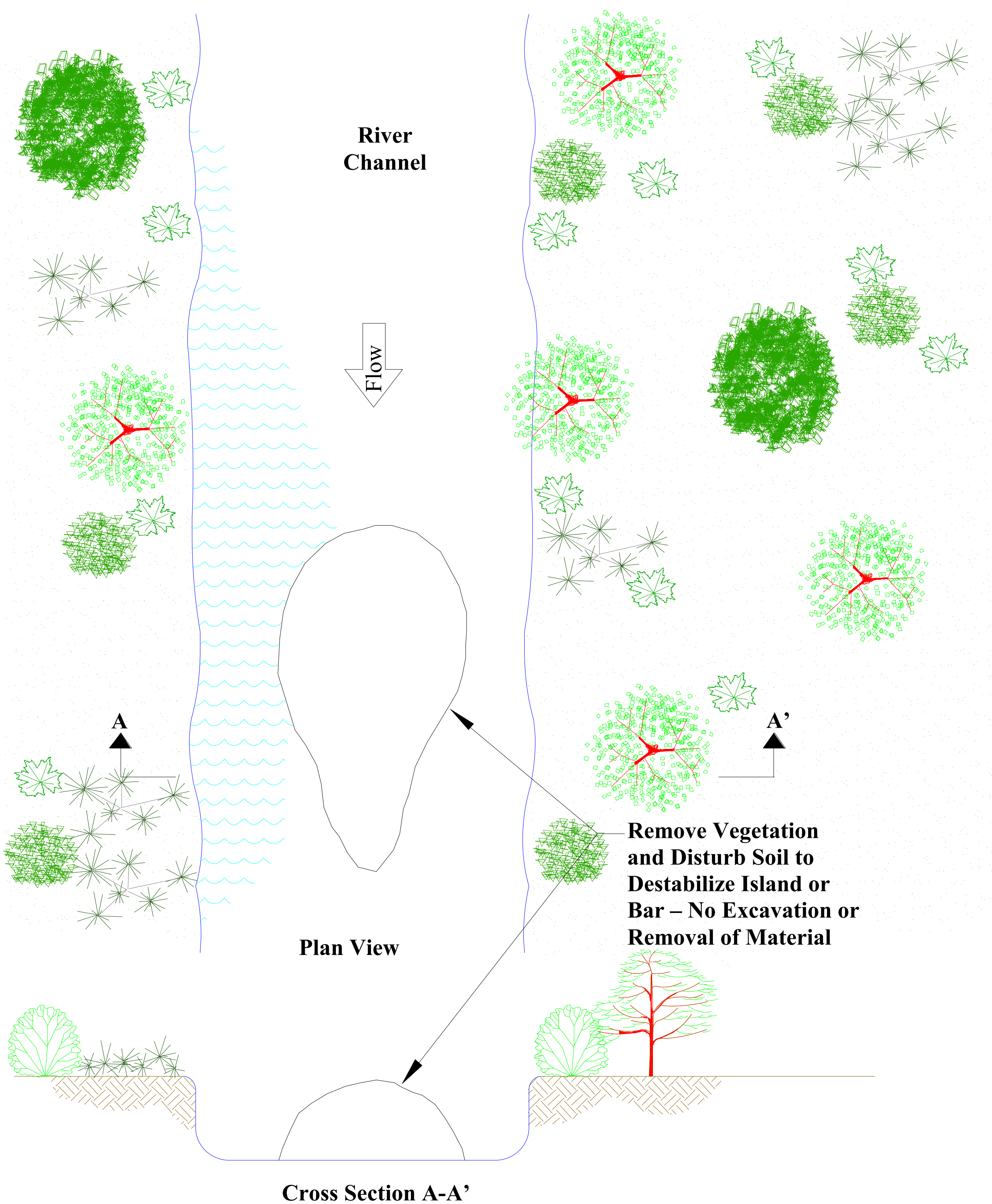

Figure 4-9: Destabilization of island bars

4.4.10 Gradient-Control Structures

Gradient-control structures (GCS) used in restoration (also called gradient restoration facilities or GRFs) are low head weirs constructed at an angle perpendicular to the main channel flow with gently sloping downstream aprons that simulate natural riffles in the channel (Fig. 4-10). Typical GCSs are built using steel sheet piling and rock (Fig. 4-11). The rock is placed at the desired slope for the GCS and retained by sheet piles driven in lines across the river at the upstream and downstream ends of the structure. Certain features are common to most GCSs including a control section for changing the grade, a section for energy dissipation, and protective measures such as berms for sections upstream and aprons downstream of the facility. There are two basic types of GCSs: bed control structures and hydraulic control structures. GCSs are generally used for regional channel stabilization when sediment loads and transport capacities become unbalanced (Hey, 1996). Generally, they are most effective with drop heights of less than 3 feet.

The GCSs are typically designed to create diverse velocities and flows across the structure and to allow passage of fish. The crest of the bed control structure is typically constructed at or near the existing channel grade upstream of the incised section channel. The primary purpose of GCSs is to stabilize or raise the riverbed. A GCS serves as an artificial "hard point" on the river that reduces or eliminates scouring, down-cutting, and entrenchment of the river channel. These structures indirectly increase channel stability above the facility. They prevent the upstream movement of an existing down-cut section of the channel (a process known as headcutting) that can lengthen the zone of channel entrenchment. In some sections of a river, more than one GCS may be needed at intervals along the main channel to achieve the desired results.

Purpose of Technique. GCSs are used to control channel degradation and incision, raise the elevation of the riverbed and upstream water surface, reduce upstream velocities, and trap finer sediments. GCSs are typically designed to reduce the energy slope in a degradation zone to a point where the stream is no longer capable of scouring the bed. They are frequently used downstream of a section that has incised and where there is a need to reestablish a desired grade to the channel. These structures enhance channel stability upstream by decreasing the slope upstream. The velocity and scouring power of the flow are therby reduced.

Considerations. The applicability of a particular structure to any given situation depends upon a number of hydrologic, sediment, and geomorphic conditions and project objectives. GCSs may affect sediment transport and, consequently, can impact downstream habitat conditions. Often, GCSs are designed to be submerged at flows less than bankfull so that the frequency of overbank flooding is not affected. However, if they exert control through a wider range of flows, the frequency and duration of overbank flows may be impacted. In this case, the safe return of overbank flows must be considered when siting a GCS because upstream out-of-bank flows that produce flooding and potential headcutting and downstream erosion could damage the structure.

As fixed points in the channel, GCSs can develop strong vortices adjacent to the bank, potentially causing instability, and thus, energy dissipation structures may be required (Hey, 1996). GCSs produce hard points in the channel that "freeze" the channel location and configuration. Consequently, these facilities should not be used where future options to promote lateral channel movement or other dynamic channel features may be desirable. Additionally, upstream changes in channel alignment may threaten GCSs or the stabilized banks downstream.

Habitat Implications. GCSs can create diversity in aquatic habitats through variable flow velocities and depths. The riffles, ponded water, and scour holes associated with a GCS would provide habitat for adult silvery minnows. GCSs may also have limited benefits related to the retention of eggs and larvae. Information on how best to construct aprons that allow minnows to pass upstream through the structure's riffles is still evolving, and monitoring of their performance will be necessary. With respect to the flycatcher, elevated water tables upstream could enhance the density of vegetation and improve nesting habitat. Provisions for silvery minnow fish passage must be considered to avoid habitat fragmentation.

Figure 4-10: Installation of a gradient control facility
(Photo courtesy of T. Bauer, Reclamation)

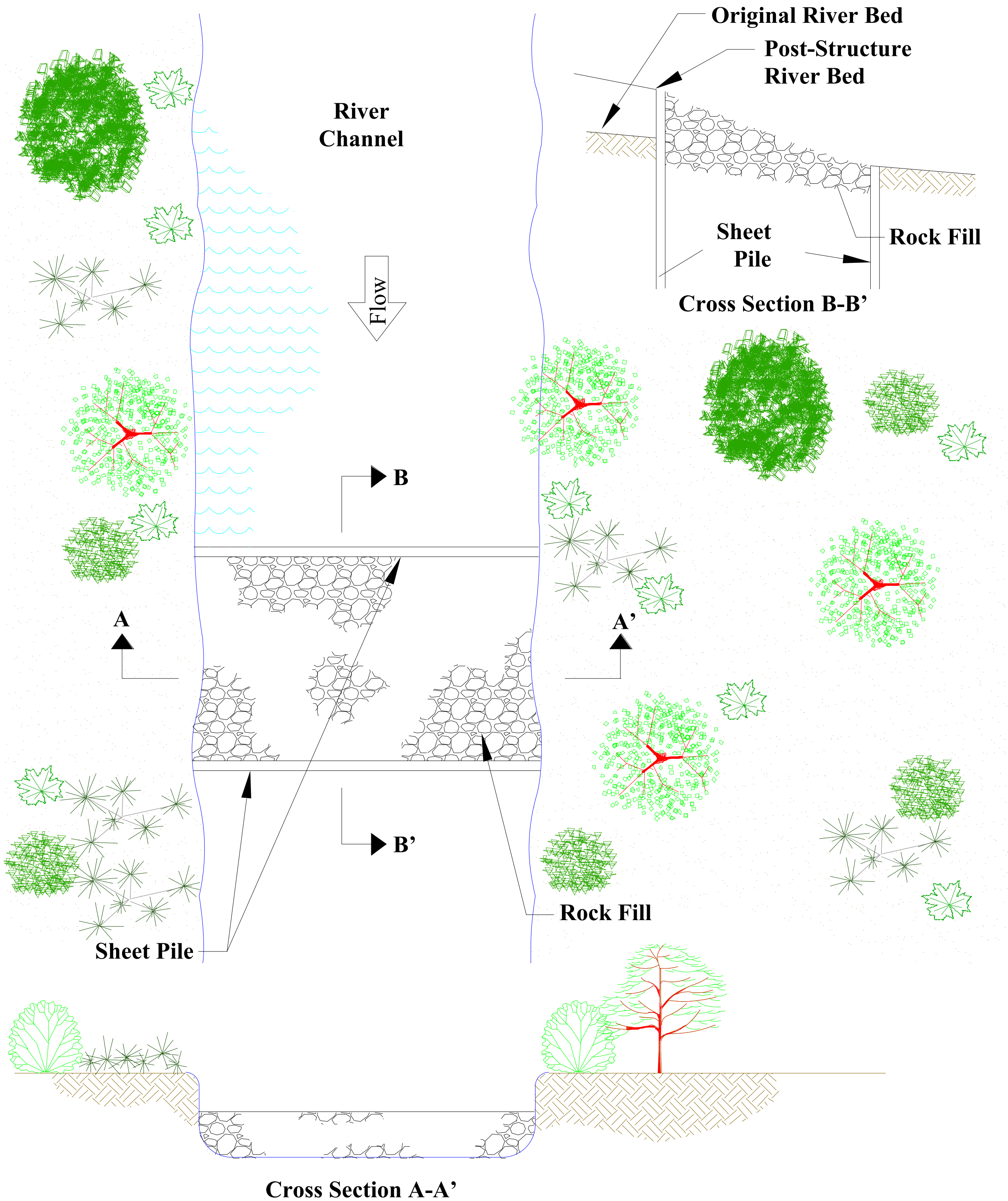

Figure 4-11: Gradient-control structure

4.4.11 Woody Debris

Woody debris includes trees, stumps, root wads, large branches, etc. in the main channel or in the water near the bank that create localized aquatic habitats (Figure 4-12). Woody debris may be anchored to the river bottom, bank or bankside trees, or placed without anchoring. Woody debris may be placed in clusters or distributed at low density along a reach of the river. Unanchored woody debris is expected to move periodically with high flows, possibly settling in relatively quiet waters downstream. Anchored debris is expected to remain in place until it decomposes.

The historic amount, distribution, and character of large woody debris (including trees and larger branches) in the Rio Grande are unknown, but it is likely that channel avulsions and bank failures resulted in significant quantities of large woody debris. Past river maintenance activities include the removal of logs and trees to prevent the obstruction or deflection of river flows (BOR, 1993). Preventative maintenance included mowing, root plowing, and herbicide treatments of bank and bar vegetation, which reduced the amount of woody debris in the river. Snagging of standing trees was frequently practiced after high-flow events, which commonly caused bank erosion and undermined trees (BOR, 1993). Combined, these past activities limited the extent of woody debris in the MRG. In recent years, snagging, bank stabilization, and other channel maintenance activities have been practiced less frequently along the MRG and are primarily limited to addressing potential hazards to levees and other infrastructure.

Purpose of Technique. Woody debris can provide a variety of benefits to silvery minnows by (1) creating localized slow-water habitats for all life stages, (2) trapping leaf-litter that provides organic substrates for organisms used as food, and (3) providing a substrate for algae. While specific information is lacking for the MRG, historical information indicates that large woody debris (snags) was a common component in other southwestern rivers. For example, the Guadalupe River in Texas reportedly contained over 1,365 snags per mile (Sedell and Beschta, 1991).

Considerations. Drift and downstream accumulations of large woody debris can affect the operation of diversion structures and the integrity of bridge crossings and pipelines. Fixing large woody debris by cabling may reduce the potential for transport. D'Aoust and Millar (2000) provide guidance for the design and construction of ballasted woody debris installations. Woody debris may initiate the formation of bars and islands, which may or may not be desirable from a river operations perspective. Introduction of large woody debris may be particularly applicable for areas where channel diversity is low and fluvial or engineering alternatives are limited. Woody debris can be used to stabilize banks in association with other restoration practices, such as embayments.

Habitat Implications. The general lack of channel structure in the MRG that provides stable substrates for algae and invertebrates suggests that introducing snags would enhance overall aquatic productivity and increase food sources for silvery minnows. Large woody debris accumulations (including fallen trees, root wads, and other mid-channel snags) provide structure for periphyton (algal) growth and the retention of drifting organic matter (Treadwell, 1999). Additionally, smaller pieces of woody debris, such as branches, sticks, and twigs on the fallen trees or captured from stream drift, create sieve-like accumulation structures that can be highly effective in trapping additional drifting materials (Gregory et al., 1991). Treadwell (1999) indicate that the more numerous and complex the array of structures, the greater the likelihood of increasing the aquatic productivity. Woody debris may also cause downstream scour forming plunge pools that could be deep-water habitat for the silvery minnow.

Increased amounts of woody debris are expected to lead to enhanced attachment sites for algal growth, increased trap structures, and better food availability and feeding habitat for the silvery minnow. Additionally, these features represent points of convergence and divergence that will promote increased diversity of flow, including net-zero velocities that would benefit egg and larvae retention and provide adult cover habitat. The placement of woody debris could promote the formation of islands or bars, which if vegetated could provide flycatcher habitat. Therefore placement of debris should be implemented following evaluation of channel characteristics and trends in each reach.

Figure 4-12: Large woody debris along river bank creates localized aquatic habitat
(Photo courtesy of M. Harvey, MEI)

4.4.12 Sediment Management

Increasing sediment supply to the river has been proposed to ensure that primary habitat elements for the silvery minnow are provided or restored. The technique may involve the addition of sediment into the system by mobilizing it behind dams (i.e., Jemez Canyon, Galisteo, and Cochiti) and allowing it to reach the river. In addition, sediment supply may be locally increased by reconnecting arroyos and disposing of spoils from restoration-related excavations, such as bank lowering projects. Sediment management issues are currently being studied by the Corps.

Sediment management is proposed primarily to address the observation (made during silvery minnow monitoring) that the slivery minnow is most commonly found in areas were the bed is predominantly silt and sand (see Section 3.1.4.3). The addition of sediment may have important consequences on channel morphology. Increased amounts of coarse material (>1.0 mm) may result in channel aggradation. Thus, sediment additions increase the potential for channel widening, channel narrowing, and the formation of bars and islands. These changes could potentially benefit silvery minnows in some reaches of the MRG, whereas in other reaches, water conveyance and/or property could be adversely affected. The chemical and physical nature of the sediments behind the dams must be evaluated prior to implementation of this practice. In particular, the potential for contaminant transport from the reservoir sediments to the river needs further investigation. Potential problems with conveyance of flood flows and safety issues require that further investigation prior to implementation of the practice on a significant scale.

4.4.13 Fish Passage

As discussed in Section 1.3, RPA R in the 2003 BO requires Reclamation, in coordination with the Service and others, to complete a fish passage at San Acacia Diversion Dam by 2008 to allow upstream movement of silvery minnows (FWS, 2003a). The RPA also requires Reclamation and others to coordinate with the Service and Pueblo of Isleta in constructing a fish passage at Isleta Diversion Dam by 2013. Section 3.1.2.2 discusses the fragmentation of the longitudinal connectivity of the river and silvery minnow habitat due to irrigation diversions and Cochiti Dam.

To facilitate development of the required fish passage structures for silvery minnows, Bestgen et al. (2003) provided a set of recommendations for their design and implementation using results from their series of laboratory and flume studies:

- Maximum velocities encountered by fish for even very short periods (i.e., 10 seconds) should not exceed about 100 cm/second at 23º C (73º F) and about 80 cm/second at 15º C (60º F).
- A mix of flow velocities in the fishway, either via a refuge, resting pool(s), boundary layer, or all of these features should be provided.
- Maximum velocities in baffled fishway slots should not exceed about 50 to 80 cm/second (ca. 20 to 30 inches/second), depending on fishway length.
- Maximum water velocities in shorter rock channel fishways should not exceed about 100 cm/second (34 inches/second), or about 75 cm/second (30 inches/second) in longer ones, provided there is substantial lower velocity boundary areas, boulder velocity breaks, channel margins, or resting pools. A channel greater than 1.25 percent is recommended.
- The inclusion of larger, cobble-sized rock in the passageway should be considered to provide a more natural array of cover where resting fish could seek refuge.
- Attraction flow velocity should be somewhat faster than the water into which it flows.
- Attraction flows should be tranquil, not turbulent.

4.5 Riparian Vegetation Restoration Practices

The habitat requirements of the flycatcher were discussed in the Section 3. This subsection introduces restoration practices that can be used primarily to improve riparian habitat for the flycatcher. The discussion includes passive and active restoration techniques that could be used either separately or in combination. In general, these practices are intended to initiate regeneration of riparian plant communities and to promote native plant species. Riparian habitat practices can also be used in combination with aquatic restoration projects. Auble (1999) suggests that restoration activities should emulate the environmental conditions created by natural disturbances, to the extent possible. Stromberg (1999) likened mechanical clearing and burning of saltcedar to the effects of large floods, but recommends such strategies be part of a larger plan to restore river processes. In many areas of the MRG bosque, fuel loads associated with accumulations of woody exotic vegetation pose fire risks; many of the techniques discussed below can also help to address this issue. The specific restoration strategies discussed in the following subsections include:

1. Removal and Control of Exotic Vegetation
2. Passive Restoration of Riparian Vegetation
3. Active Restoration of Riparian Vegetation
4. Hydromodification
5. Wetlands

4.5.1 Removal of Vegetation and Control of Invasive Exotics

Woody exotic species, primarily saltcedar and Russian olive, dominate many areas of the MRG, replacing and displacing native woody species (predominantly willow and cottonwood). Exotic plants may exacerbate basin depletions and increase fire hazards. Bosque degradation may be reversed locally by removing the undesirable exotic species to reduce fire hazards that can destroy flycatcher habitat. Control and removal of woody exotics on a restoration site would typically be required prior to reestablishing native plant communities whether using passive or active restoration techniques. Water salvage projects that convert exotic-dominated stands into lower-water-use plant communities may require exotic plant removal prior to planting.

Several treatment options are available to control and remove exotic vegetation and prepare the site for revegetation. Mechanical removal, prescribed burning, chemical control, biological (invertebrate and grazing) control, and flow regulation are all methods that have been used either singly or in combination to control invasive exotics. Because saltcedar, Siberian elm, and Russian olive have the ability to re-sprout following treatments, control of these species typically requires multiple treatments.

Considerations. The selection of appropriate methods to control exotic species is site-specific, depending on the site's potential to support flycatcher habitat and native species, the proximity to the water table or open water, the density and uniformity of exotic species in the stand, and the presence of obstructing features like jetty jacks. It is also important to recognize that with any control practice, ongoing maintenance will probably be required to keep saltcedar and Russian olive from reinvading.

Habitat Implications. Removal may benefit flycatcher habitat by reducing the potential for catastrophic fires. Removal efforts focused on exotic species must balance concerns for present and future flycatcher habitat. Small, local activities to remove saltcedar and Russian olive are not anticipated to significantly increase water flow in the river to the point where it would benefit silvery minnow conservation or recovery goals.

Costs. Costs for control and removal of exotic species are difficult to ascertain given that such activities are done in combination with other methods, depending on site-specific conditions, and that it may require several years to achieve control. Many published cost estimates for control do not fully account for either labor or equipment. Additionally, many cost estimates are specific to one phase of a total treatment (e.g., herbicide application) and may not account for follow-up treatments (e.g., prescribed burn, skidding) that are needed to effectively clear the site for revegetation. Costs also often do not include preliminary site surveys or follow-up monitoring activities. Typical costs for individual strategies are provided in their respective section below and are further summarized in Table 4-2, with the understanding that they are provisional.

The following sections provide an overview of exotic plant removal and control methods and their potential effectiveness, benefits, and disadvantages. Table 4-2 summarizes some important aspects of the removal practices.

Table 4-2: Removal and control methods for woody exotic species

Method	Effectiveness	Advantages	Disadvantages	Costs	Where	Remarks	Citations[1]
Mechanical (bulldozer, root-plow, mowing, raking, grubbing, etc.)	Can be low the first year, up to 99% with subsequent treatments	High level of control Less likely to damage non-target vegetation Fastest potential for plantings	Labor intensive More expensive than herbicides Erosion could be an issue	Bulldozers, root plow and raking $600-$800/acre; flail/rotary mowers $800-$1500	Monotypic stands of mature saltcedar	Root plowing most effective in the hottest part of the year Fall/winter above-ground treatments avoid impacts to nesting birds Waste disposal an issue	Zouhar, 2003 Carpenter 1998 Pers. Comm T. Caplan
Hand crew (cutting, chipping, herbicide)	Up to 99% after 2-3 years of follow-up herbicide on re-sprouts	Less likely to damage non-target vegetation Less ground disturbance	Labor intensive Longer treatment period	$1,800-$3,000/acre (non-inmate crews)	Small or mixed native stands or sensitive areas, where don't want excessive soil disturbance	Can produce firewood	Pers. Comm. Y. Najmi M. Schmader
Mechanical with mulching and herbicide	80 - 95% without herbicide on re-sprouts, 99% with follow-up treatment(s)	Shorter treatment period, less expensive c.f. hand crew.	Some hand work required around natives and jetty jacks Potential impact on non-target vegetation	$1,200-$2,500/acre	In areas not safe to burn	Costs can be reduced by leaving more slash on site Ground disturbance may stimulate native plant regeneration	Pers. Comm. Y. Najmi M. Schmader M. Reynolds
Mechanical/Herbicide (mechanical treatments followed by herbicide treatments to stumps, foliage or stems)	Highly effective in 2-3 years of follow-up herbicide on re-sprouts	More effective control than mechanical treatments alone	Potential impact on non-target vegetation	Not available	Monotypic and mixed stand, where don't want excessive soil disturbance	Waste disposal an issue	Caplan, 2002
Herbicide	See Table 4.3						
Herbicide/Mechanical (aerial spraying followed by mechanical removal of dead biomass)	80 - 95 %	Less labor. Less expensive	Potential impact on non-target vegetation	$500+/acre	Monotypic stands, where burning is not safe	Waste disposal an issue	Taylor et al., 1990

Table 4-2 (continued): Removal and control methods for woody exotic species

Method	Effectiveness	Advantages	Disadvantages	Costs	Where	Remarks	Citations[1]
Herbicide/Burn (aerial spraying followed by burning standing dead biomass)	High, up to 96%	Less labor Less expensive	Potential fire danger Potential impact on non-target vegetation Herbicides must cover the entire plant	$113/acre aerial spraying	Large monotypic stands, where burning is safe	More appropriate for larger areas No waste disposal issues Effectiveness of basal spraying unknown Air quality issues	Carpenter, 1998
Prescribed Burn (slash pile or ground burn)	High, up to 99%	Time-efficient means to remove large amounts of biomass	Potential fire danger Resprouting can occur without herbicide Potential impact on non-target vegetation	up to $500/acre	Slash pile removal or ground burn following fuels reduction, limited use in native stands	Following herbicide treatment, wait at least 2 weeks before slash pile and ground burns Air quality issues	Pers. Comm. D. Boykin
Burn/Herbicide (burning standing biomass followed by herbicide applications to root-sprouts)	30% (saltcedar) without follow-up herbicide, >95% with follow-up treatment		Potential wildfire danger Potential impact on non-target vegetation	Unknown	Monotypic exotics that are well contained	No waste disposal issues Air quality issues	B. Racher
Biological Control - Arthropods	Moderately high, up to 85%		Potential ecological ramifications Not available	Unknown	Not within 200 miles of flycatcher habitat	Experimental Currently not available due to flycatcher habitat concerns	DeLoach, 1997 Gould, 2002
- Herbivores	Unknown, assumed moderate to high	Safe, low labor	Potential impact on non-target vegetation	Unknown	Mixed to monotypic stands, to get re-sprouts	Requires fencing May need to repeat several years	Zouhar, 2003 Carpenter 1998

[1] Personal communications: Y. Najmi (MRGCD): M. Schmader (City of Albuquerque Open Space Division); M. Reynolds (Valencia Soil and Water Conservation District - Non-Native Phreatophyte Removal Program); D. Boykin (New Mexico Forestry Division - Socorro District); T. Caplan (Parametrix) and B. Racher (Ranger Resource Management).

4.5.1.1 Mechanical

Mechanical removal can be used selectively to eliminate exotic plants within stands of mostly native vegetation, or to remove large swaths of exotic vegetation where few or no native plants are present. Mechanical removal of exotic vegetation includes the use of flail-mowers, skidders, chippers, bulldozers (sometimes equipped with root plows, chains or rakes), extractors, mulcher-grinders, and root-plows to remove exotic woody plants and their root systems. Hand crews may be used in some instances to clear vegetation.

Mechanical removal is typically a four step-process: (1) aerial stem removal; (2) slash disposal; (3) root plowing and raking; and (4) root crown disposal. Often this process is repeated to achieve more effective control. A bulldozer or rotary brush cutter removes aerial stems. The slash material is disposed of by burning, chipping or removing from the site. Roots are then plowed and raked or grubbed if cottonwoods are present, and piled. The final step is the disposal of the root wads. With follow-up treatments, mechanical removal can be 99% effective. Depending on the equipment used, treatment costs may run from \$600 to \$1,200 per acre. Jetty-jacks or other structures in the floodplain may complicate the use of heavy equipment.

4.5.1.2 Herbicides

Herbicides have effectively controlled woody exotic species when applied correctly. Herbicide applications minimize soil disturbance and can provide effective kill rates. Three herbicides are effective in the control of both saltcedar and Russian olive: triclopyr ester (e.g., Garlon 4®), triclopyr amine (e.g. Garlon 3®), and imazapyr (e.g., Arsenal®). Glyphosate (e.g., RoundUp®) has also been reported to kill mature Russian olive trees (Parker and Williamson, 1996). As with all herbicides, instructions should be followed closely and care must be taken in their handling, especially near open water and wetlands. In New Mexico, a commercial license for herbicide application is typically required.

Three different herbicide techniques can be employed to control saltcedar and Russian olive: foliar application, the cut-stump method, and basal-bark application (Frost, 2003). Table 4-3 summarizes the considerations relative to these chemical treatments.

<u>Foliar application</u>: Imazapyr is the only effective herbicide against saltcedar in foliar applications. This chemical is nonselective and best suited for saltcedar monocultures where total removal of vegetation is desired. Duncan and McDaniel (1992) suggest that foliar treatment should focus on young stands and in areas with saltcedar densities of <400 plants per acre. Studies have shown that aerial spraying of imazapyr provides 80-95% control of saltcedar; similar control was achieved with tank mix applications of imazapyr and glyphosate (Duncan and McDaniel, 1992; Taylor, personal communication). In most situations, it is recommended that the stand be left undisturbed for 2 to 3 years to ensure that the systemic chemical fully kills the plants (Johnson et al., 2002). Aerial spraying generally requires fairly large sites, and depending on the equipment used, costs can approach \$200 per acre. Additional costs would result from subsequent vegetation removal and disposal and, potentially, for follow-up herbicide applications.

Timing of foliar applications may have a strong influence on the percent kill for both Russian olive and saltcedar root sprouts and should be considered when planning treatment activities. For example, foliar treatments on Russian olive root-sprouts may be most effective with early to midsummer herbicide applications using either glyphosate (5% solution) or triclopyr (25% solution) (McDaniel et al., 2002a & b). Conversely, foliar treatments on saltcedar may be most effective using either imazapyr (1% solution) or triclopyr (25% solution) during late summer months only (McDaniel et al., 2002a).

<u>Cut-stump</u>: Triclopyr is typically the chemical used for cut-stump and is most effective for larger mature trees (Tu, 2003). This method is recommended when desirable species occur in the treatment area. This method is a combination of mechanical and herbicide treatments performed by hand crews using chainsaws and backpack sprayers. For the cut-stump method, stems are cut within 2 inches of the ground surface and herbicide is immediately applied with a brush or backpack sprayer to the entire circumference of the stem cambium (Frost, 2003; Tu, 2003). Johnson et al. (2002) recommends full-strength herbicide application for saltcedar, though lower concentrations have been shown to be effective (Carpenter, 1998; Parker and Williamson, 1996). For Russian olive stumps, effective control can be achieved using a 50% triclopyr solution (Caplan, 2002). It is recommended that with either species, follow-up foliar treatments of root-sprouts begin within 12 months (Carpenter, 1998) and continue for at least 2 years (Caplan, 2002). The cut-stump method is labor-intensive and can be particularly difficult in dense stands where movement is limited. Mortality with the cut-stump method can reach 95% and it may use much less herbicide than foliar applications (Frost, 2003). Triclopyr applications are recommended in the fall to facilitate its translocation into plant roots. An alternative technique to the cut-stump method involves using a hatchet to make several cuts into the plant's xylem about 4 feet from the ground and spraying herbicide into the cuts (Parker and Williamson, 2003).

Table 4-3: Considerations and recommendations for chemical treatments of woody exotics

Consideration	**Treatment Methods**		
	Cut-stump	**Basal Bark**	**Foliar Spray**
Plant Stage	All stages, triclopyr in summer and fall.	All stages, but most effective when applied to smooth bark portions of stems <3" in diameter. Apply to saltcedar during dormancy. Success in Russian olive may be throughout the year.	Best results occur with an aerial application of imazapyr in the late summer to early fall, until dormancy begins.
Treatment Process	Immediately paint stumps with triclopyr. Use a water-soluble dye to track the treated plants.	Spray the lower uncut 15" of the plant with triclopyr in an oil carrier. Spray the entire bark surface of the stem.	Herbicide and wetting agent are applied via ground-based sprayers (ATVs or trucks) or aircraft.
Herbicide Application	Thoroughly treat each stump, especially the cambium layer just inside the bark. Stumps must be wetted completely without runoff for good control.	Low-volume application: mix 25 to 30 gallons triclopyr with oil to make a 100-gallon mixture. Applied to plants with stems less than 3" in diameter. Inconsistent results.	Imazapyr is applied with proper surfactant until the saltcedar is wet, but not dripping. The crown and roots of large trees should not be disturbed for 2 years to allow imazapyr to move throughout the tree to prevent re-sprouting.
Effectiveness	Most popular and effective in areas unsuitable for aerial or ground rig applications. Used near water to avoid drift and contamination of water.	Retreatment of the stems that were not killed is difficult compared with the cut-stump method.	Effective on large stands with few non-target plants growing among the saltcedar. The shoots normally die within 1 year, the roots within 2 years.
Retreatment	Is necessary to clean up missed stumps and re-sprouts.	May need to re-treat the following years.	If necessary

Adapted from Johnson et al., 2002

Basal bark application: This technique is most effective on small diameter trees (<3 inches) with smooth bark (Johnson et al., 2002; Parker and Williamson, 1996; Tu, 2003). When spraying larger trees with thick, rough bark, the spray should extend up to include some smooth bark and even then the method may only provide 50% control (Parker and Williamson, 1996). The entire circumference of the stems must be treated with triclopyr mixed with methylated seed oil to be effective (Johnson et al., 2002).

Again, dense stands can make spraying the basal portions of the trees difficult. While this method eliminates the need to precut the plants, it requires up to 5 times as much herbicide as the cut-stump method and results in lower mortality (Frost, 2003). The most effective time to apply the herbicide is from May through September for Russian olive (Parker and Williamson, 1996). Johnson et al. (2002) recommend spraying dormant saltcedar.

Use of herbicides in conjunction with other techniques may be the most effective means of controling exotic species. McDaniel and Taylor (1999) found herbicide/burn treatments to be less expensive and more effective than mechanical methods alone for dense uniform stands of saltcedar. Other combinations such as mechanical treatments followed by spot or carpet-roller herbicide applications to control root-sprouts may also be an effective strategy.

4.5.1.3 Prescribed Burning

Fire can be used as a technique for removal or disposal of exotics following the use of some other removal technique, such as herbicide/mechanical. Fire is also a useful tool for disposal of slash and deadwood that has been consolidated after use of other removal techniques (i.e., herbicide application or mechanical removal). Prescribed burns could also prove effective to clear monocultures of either live exotic vegetation or dead standing woody material left following herbicide application. Fire can also be used to maintain understory fuel loads after fuels reduction. As with aerial applications of herbicide, fire should be used cautiously when desirable vegetation is present.

Prescribed burns may be more suitable in rural areas of the MRG and should be done under weather conditions that minimize impacts to native vegetation and neighboring property. However, the weather conditions that may lessen impacts to native vegetation may also result in poor air quality. Potential use of fire needs to be evaluated against the potential loss of wildlife or desirable vegetation to determine if it is worth implementing. This technique should not be used in areas containing active foraging and nesting habitat for flycatchers. Also, ash residue washing into surface waters following a fire may impact silvery minnows directly by clogging gills; due to its mobility in river environments, ash would likely have minimal impacts on other aspects of minnow habitat. Sites with high loads of dead-and-downed fuel and ladder fuels will require careful fire planning or potentially the selection of another method to clear exotics. Controlled burns are not appropriate during flycatcher nesting season. Typically, fire may be best applied during cooler times of the year (November to March) and following rains to help contain the treatment area.

Fire alone should not be considered as a saltcedar control method, as saltcedar usually survives all but the hottest fires. Without employing other techniques (e.g., herbicide, root-plowing) saltcedar will re-sprout from unburned root systems. Prescribed burning followed by herbicide applications has been shown to be up to 99 percent effective in preventing sprouting (USFS, 2003). Herbicide can be applied to saltcedar root-sprouts that regenerate following a fire with a rotary brush or a basal application.

4.5.1.4 Biological Control

These techniques include use of insect control agents, such as the leaf-feeding beetle, *Diorhabda elongata*, and the mealy bug, *Trabutina mannipara*. They also include control by herbivorous animals, especially livestock (e.g., goat browsing). These techniques may be appropriate in areas where mechanical removal and/or burning could be used at a later date to eliminate standing dead wood, if revegetation is a goal. If revegetation is not a goal, they might be appropriate where water salvage is desirable, thus benefiting silvery minnows more than flycatchers (the standing dead wood would not benefit flycatchers). Both techniques are discussed below.

Arthropods: Nearly 200 insect species in China and the former Soviet Union have been identified as natural enemies of saltcedar (Stelljes and Wood, 2000). Fifteen of those insects have been investigated by the U.S. Department of Agriculture as potential biological controls to reduce saltcedar populations. There are no known biocontrol agents that are selective for Russian olive (Tu, 2003). The leaves of Siberian elms can become completely skeletonized by the imported elm leaf beetle (*Pyrralta luteola*) and its larvae, though it is uncommon that such attacks damage more than a branch (Cranshaw and Zimmerman, 2003).

Two insects, a mealybug (*Trabutina mannipara*) from Israel and a leaf beetle (*Diorhabda elongata*) from China, have preliminary approval for release for saltcedar control. Gould (2002) reported that they have been released on a limited basis in nursery and field cages in the western United States. The leaf beetle and its larvae, once established, eat only saltcedar leaves and rapidly defoliate the plant. The mealybug was collected near the Dead Sea and may have limited applications in the MRG because it is not adapted to areas that freeze. Biological control could reduce saltcedar abundance by 75-85% in 5 to 10 years with the introduction of these and other biocontrol agents, though the degree of control will likely vary in any given area (DeLoach, 1997).

Vegetation control by introduced arthropods is still largely experimental, so there are concerns about what impact the released agents might have on existing vegetation. Research into the effectiveness of the technique is underway (Gould 2002). Their use should not be implemented until this research is completed.

Since the 1995 listing of the flycatcher, plans to release these insects have been put on hold in the MRG because it is presently or may potentially become flycatcher habitat. The primary concern with the beetle is the fact that the rate of defoliation is unknown and could be faster than the natural rate of willow recruitment. The listing restricts the introduction of biological controls (beetles) within 200 miles of occupied flycatcher habitat. As such, this method is unavailable for saltcedar control in the MRG at this time.

Herbivores: Grazing is generally thought to promote the spread of woody exotics because livestock prefer to consume native willows and cottonwoods, making room for the invasive trees. However, cattle, sheep, and goats will consume saltcedar and Russian olive when other, more desirable forage is unavailable. Highly managed, intensive grazing by goats has been demonstrated on a limited basis (S. Grogan, MRGCD, personal communication.).

Livestock have been observed peeling bark from tree trunks, browsing on root-sprouts, and eating the foliage of both saltcedar and Russian olive. Saltcedar is presumed to have low forage quality (Zouhar, 2003), and consequently, it will not be grazed if other more palatable plants (cottonwoods and willows) are available (Dick-Peddie, 1993; Stromberg, 1997). Because saltcedar is high in tannins, digestion of it is often inhibited in livestock (Zouhar, 2003). Goats appear to metabolize this plant better than sheep or cattle. Wildlife and livestock are occasionally known to browse on Russian olive, since its forage quality is higher than that of saltcedar. Controlled grazing may be more effective on re-sprouts following mechanical treatments or prescribed burns (Carpenter 1998) and on young growth rather than on large mature trees. Grazing on woody exotic vegetation has not been fully proven to be a practical control technique, but the practice has potential value in small-scale applications were livestock can be intensively managed.

4.5.2 Passive Restoration of Riparian Vegetation

Passive restoration of riparian vegetation involves letting volunteer vegetation establish on disturbance sites rather than planting, particularly following an overbank flood event. This approach may be selected following an active disturbance such as bank lowering or fire. In other cases, passive revegetation will occur on natural disturbance areas (e.g., bars and islands). Areas that are not maintained by frequent flooding, like bank-line embayments and arroyos, will also revegetate over time without intervention. Passive restoration of riparian vegetation may also include more active components such as non-native plant removal and site preparation, flow manipulation, livestock management or bank lowering. The initial composition of the vegetation and the trajectory of the plant community are determined by a complex set of factors. A more detailed discussion of the likely vegetation trajectory associated with passive vegetation is included in Section 3.3.

As discussed above, passive restoration is most simply defined as allowing the river to perform functions that previously occurred to create and maintain quality habitat areas. For example, spring floods may introduce water into the floodplain and provide opportunities for recruitment of native vegetation. This process might be done under current river operations or with some modifications to the flow regime. In the southern reaches, moderate flows (e.g., 3000 to 5000 cfs) may be sufficient to establish new age classes of native vegetation.

Purpose of Technique. The purpose for selecting a passive revegetation approach for sites that have been actively disturbed is to reduce costs and promote a diversity in species structure and density that is difficult to achieve with plantings. Passive revegetation can also be employed as part of other restoration projects, in particular those that enhance aquatic habitat (e.g., bank lowering, channel widening). Otherwise, passive colonization of bars and islands will occur in the absence of interventions. Throughout the MRG, islands as well as point and channel bars tend to develop readily. Past channel dynamics and maintenance activities worked to keep these areas clear of vegetation, increasing their mobility during high stages of the river. For significant areas of the floodplain to revegetate passively, removal of jetty jacks and other lateral constraints could be required to allow the channel to move within the levees and promote greater diversity of both aquatic and terrestrial habitats.

Considerations. Flow manipulation and altering the magnitude, timing, frequency or duration of floods also could enhance vegetation recruitment. The flow requirements that would promote native species regeneration have been studied on the MRG, but reach-specific requirements for successful implementation of this technique would need to be clarified and tested. Potential changes in flow management alternatives would need to be evaluated, with the feasibility and benefit of this technique analyzed within the existing statutory constraints on river operations. Under current hydrologic and species composition conditions, the efficacy of passive revegetation in recruiting native species is not well understood. Vegetation surveys of recently formed bars and islands in the Albuquerque reach indicate that mixed stands of native and non-native species develop under a passive revegetation approach (Section 3.3; Milford et al., 2003).

Passive restoration of flycatcher habitat would likely focus on sites where barren substrates are prepared or available and where there is a water table sufficiently near the surface to support dense, willow growth. Site conditions (groundwater fluctuation and depth, soil texture and chemistry, browsing pressure, etc.) would need to be evaluated as well. Livestock grazing needs to be managed to allow vegetation establishment and to protect potential flycatcher breeding and fledging activities against destruction or trampling effects. As with any revegetation effort, undesirable species can co-establish because of the overlap with native plants in their requirements (Auble, 1999). In cases when the desired plant composition is not achieved passively, intervention (e.g. selective herbicide, disking, grazing) may be necessary to reinitiate plant succession.

Habitat Implications. Passive revegetation that results in dense willows or mixed stands with exotics could provide suitable flycatcher habitat for breeding, feeding, and/or migration.

4.5.3 Active Restoration of Riparian Vegetation

Active restoration of riparian vegetation involves planting, seeding, or water management activities aimed at establishing a selected plant community. Selective planting strategies like seeding and transplanting into established stands could also be considered in areas to enhance the plant species diversity without impacting established plant communities. Habitat restoration activities are likely to be part of comprehensive projects where exotic species have been removed intentionally or by a wildfire. Moreover, revegetation projects may be applicable to sites designated for water salvage or to prevent noxious weed encroachment.

Considerations. Habitat restoration efforts for flycatcher habitat would likely focus on sites that have the greatest potential to recruit breeding pairs and to support native riparian species. General considerations for active restoration practices are similar to those for passive restoration activities, except that the ability to determine the composition of the resulting plant community is increased with active techniques.

Habitat Implications. Dense plantings of willows alone, or with a light to moderate overstory of cottonwood, could provide potential increases in suitable flycatcher habitat. Open meadows and wetlands could also improve overall riparian community diversity, which may benefit habitat conditions for flycatcher foraging. Revegetation and stabilization of banks and erosion control may impact minnow habitat depending on the site-specific conditions along the river.

4.5.3.1 Direct Seeding

Broadcast and drill seeding has been used extensively across the western United States, particularly in dryland reclamation projects. However, drill or broadcast seeding of willow or cottonwood is not recommended because the small seeds have a limited viability period. Willow and cottonwood seed can be collected and propagated in the greenhouse when cutting stock is limited or preservation of local genetics is desired (Dreesen et al., 2001). Thus, direct seeding is primarily limited to grasses, forbs, and some shrubs.

Before direct seeding, site preparation may be required to improve the seedbed and remove undesirable weeds. Disturbance close to the channel may decrease bank stability (Platts et al., 1987). Jetty jacks and other structures could also hamper seedbed preparation and drill seeding. In such cases, broadcast or hydroseeding may be appropriate. Seeding, compared to the planting techniques described below, is the least expensive on a unit area basis and may have general applicability for establishment of grasslands for water salvage projects.

4.5.3.2 Pole and Whip Planting

Pole planting is a technique to rapidly establish new growth of cottonwood and willow in areas where these species are lacking or underrepresented due to exotics competition or as a result of clearing of exotic vegetation or fire. The technique essentially circumvents the rather tight environmental requirements for cottonwood and willow seed germination and jumpstarts vegetative growth. It is used where natural recruitment from seeds and overbank flooding is not possible, but where the water table is high enough to support these plants once established. Although appropriate for cottonwood and willows, this technique has limited success with other native woody plants. Vegetative propagation can facilitate the quick establishment of native woody plants and help limit the potential dominance of exotic species in riparian restoration projects. Dormant cuttings of young woody materials have been used extensively for this purpose along numerous streambank bioengineering and riparian rehabilitation projects (Ogle et al., 2000; Bentrup and Hoag, 1998; Allen and Leach, 1997).

In the Southwest, pole and whip plantings have been used successfully on a local level to establish cottonwood and willows (York, 1985; Swenson and Mullins, 1985; Barron, 1996; Dreesen et al., 1999). Pole planting is typically limited to cottonwood and Goodding's willow in the MRG. Bundles of coyote willows, about five cuttings or whips per bundle, can be installed horizontally as wattles or cigar-shaped fascine embedded into a slope. Similar bundles of willow whips can also be placed vertically in a long trench or in individual holes.

Most pole and whip plantings typically follow other restoration activities such as exotic species removal or aquatic habitat improvement. Consistent water table conditions have the greatest potential effect on survival. In addition, floods that inundate new cottonwood poles for longer than 3 weeks can appreciably reduce success (Swenson and Mullins, 1985). Some sites with excessively sandy or gravelly soils may not be suitable for the cottonwood poles if the holes readily collapse. Failure of cottonwood pole plantings in the Pecos River has been attributed to groundwater salinity levels that exceeded 6.25 dS/m (Swenson and Mullins, 1985). Hoag et al. (2001) suggests that natural stands of willow and cottonwood have low to medium tolerance for soil salinity, but no threshold that may impact pole and whip plantings has been established. Higher mortality of willow whips and cottonwood poles has also been observed when they are placed too far (> 2 feet) above the growing-season water table (Swenson and Mullins, 1985; Conroy and Svejcar, 1991).

Protocols for preparing and planting poles and whips are described by Dreesen et al. (2001). Details regarding willow bundles and their installation are provided by Betrup and Hoag (1998). Cottonwood and willow cuttings are usually taken in winter prior to budding; planting is generally done in late winter and early spring. Dormant cuttings should also be kept wet and chilled until planted. Local pole and whip cuttings, when available, are important to preserve genetics. Target densities of approximately 100 cottonwoods per acre cost $2,500 to $4,000 per acre including labor (Dreesen et al., 1999). Significantly higher densities of willows would be required for flycatcher habitat and should be determined on a site-specific basis. Additionally, exclusion or deferment of grazing as well as protective measures against beaver have improved cottonwood and willow pole plantings (Vincent, 1996; Stromberg, 1998b; Conroy and Svejcar, 1991).

4.5.3.3 Containerized Stock

Containerized stock consists of nursery-grown native trees and shrubs, sapling size or larger, that have a well-developed root system that allows them to survive with little or no maintenance after planting. The stock can be easily transported to the revegetation site in plastic or biodegradable pots or burlap wraps for planting. Generally, this stock includes species of woody plants that are not easily established via pole planting or seeding. Containerized stock can be used to establish shrub vegetation that forms the understory of established riparian forest and that contributes to overall native plant diversity in cottonwood-willow bosque. Selection of container type, dimension and materials is often an economic decisions based on site-specific conditions, availability, and project goals.

Under favorable climatic conditions, transplanting rooted plants is more successful than direct seeding in small-scale restoration projects. Transplanting offers more control than seeding in that plants can be placed where they are more likely to survive. Containerized stock is likely to be used in combination with exotic species removal and pole planting as the plants can increase habitat diversity locally in a restored riparian or wetland plant community. In particular, transplants may give a competitive advantage to native shrubs in the understory and help exclude exotic species.

It is recommended that tallpot (30") containerized plants be used because they produce extensive root systems that often require less maintenance and have higher rates of survival (Dreesen et al., 2001). Tallpots appear to work especially well for understory shrubs like New Mexico olive *(Forestiera neomexicana*), and skunkbush sumac (*Rhus trilobata*). Other shrubs that may have beneficial uses include wolfberry (*Lycium pallidum*), seepwillow (*Baccharis* species), and false indigo bush (*Amorpha* species). To preserve local plant genetics, stock should be selected and grown from the seed or cuttings collected near the restoration site. As with dormant willow cuttings, plant densities should be determined on a site-specific basis. With rhizomatous plants, Hoag (2001) recommends spacing plants every 1.5 ft^2 (~20,000/acre), allowing the interspaces to fill in over one growing season. Fall is the best time to transplant to encourage the development of a strong root system. Early spring plantings can be successful in areas that may be naturally (groundwater) or artificially irrigated.

The use of containerized stock is limited to some degree due to the high costs associated with growing and transplanting the materials (Table 4-4). Labor costs associated with hand-planting containerized stock depend on the plant species, pot type, and site conditions, but they are usually equal to the cost of the plant (Hoag, 2000). Moreover, larger projects may require an extended lead time to ensure that sufficient quantities of stock are available. Due to the high costs, this method is most appropriate for smaller sites where conditions are optimal for survival.

In exceptionally dry years or for sites that do not receive periodic overbank flooding, supplemental irrigation (drip or surface) or amendments (i.e., hydrosilica) may be necessary to improve survival. Such practices will also increase associated restoration costs. Containerized stock can also be planted in small ditches cut in the floodplain (Dello Russo, 1993) or placed in topographic depressions that may catch additional water from surface runoff. As with pole and wattle plantings, the area should be protected from beavers and grazing animals.

Woody plants that become established can contribute to the overall diversity of native vegetation, both in terms of species as well as structure (height and density), and can create habitat that is attractive to flycatchers for foraging. Containerized stock may help stabilize floodplain soils in areas that have been cleared and shade out exotic plants that may re-sprout.

4.5.3.4 Supplemental Irrigation

Supplemental irrigation includes diversion of surface water to flood irrigate, periodic drip irrigation or sprinkler irrigation, as well as hand application of water to individual transplants. These activities could be employed to ensure the success of plantings. Seed germination and plant establishment could benefit from supplemental irrigation in areas that lack the potential for natural overbank flows. Once irrigation is discontinued, however, plant survival will depend either on incidental precipitation or the plants ability to access groundwater. Thus, matching plant requirements to the physical constraints of the site is essential.

Opportunities exist to use this technique as a stand-alone practice, particularly in the diversion of water to flood irrigate an impounded area and help cottonwood seed germinate. The seed could be from natural seed fall or be manually scattered on the surface. Micro-irrigation following natural seedfall has been proposed for areas where the opportunities for flood irrigation or pole plantings are limited (Dreesen et al., 1999) and as a means to preserve the genetics of a local population (Friedman et al., 1995). Friedman et al. (1995) found that on disturbed sites (water table within < 1.1 m of the surface), daily overhead irrigation was necessary for both broadcast and naturally-seeded cottonwood to germinate and establish. In the same study, willow germination was extremely low, perhaps because irrigation did not coincide with willow seed dispersal and/or it was inadequate to support root growth to the capillary fringe of the water table (Friedman et al., 1995). Dreesen et al. (1999) estimated costs for irrigating for 2 years at $2,500/acre, with the most substantial portion going to weed control and irrigation operations.

When existing infrastructure is present, flood irrigation could be used to support transplants and pole plantings or to prepare a moist surface for willow and cottonwood seeds to germinate. Such activities must provide appropriately timed flooding to minimize saltcedar germination.

Table 4-4: Restoration techniques for riparian vegetation

Method	Advantages	Disadvantages	Costs	Remarks	References
Direct Seeding	Low cost	Limited ability to target species to specific sites Not effective for cottonwood and willow	$300-3,000/acre including labor depending on seed mix	Moderately effective method for dryland reclamation	Hoag, 2000
Pole and Whip Planting	Relatively quick vegetation establishment Can out compete woody exotics	Availability of quality pole material	$3.75-12.00/pole Installation costs typically range between $15 and $38/per pole $2.00-4.00/whip bundle	Cost of pole planting is influenced by soil texture, depth to water table, and whether activities are implemented by management staff or contractors Poles and whips must extend to the capillary fringe above the water table	Dreesen et al., 1999
Containerized Stock	At high densities they may exclude undesirable plant species	Expensive	Depends on species, pot size/type, and site conditions Installation costs are typically equal to plant cost	Used for species that do not root well in the field	Hoag, 2000
Supplemental Irrigation	Improve seed germination Increase survival of transplants	Can encourage weeds and exotic species	$2,500/acre	May require soil disturbance prior to natural seedfall	Dreesen et al., 1999
Passive Restoration	Low cost	Can't impact plant community composition May encourage exotics	Unknown	Effectiveness unknown	

4.5.3.5 Hydromodification

Hydromodification is the manipulation of a site's hydrology through spring overbank flooding and controlled drawdown to enhance native willow and cottonwood seedling survival and growth (Sprenger et al., 2002; Taylor et al., 1999; Horton and Clark, 2001). Under natural conditions, survival of cottonwood and willow seedlings is contingent upon a hydrologic regime that moistens the seedbed prior to germination and supplies water to roots as they elongate to the capillary fringe (Section 3.3.4.3). If inadequate antecedent water or storage capacity of riparian soils combines with excessive stage-controlled groundwater declines, elevated water stress or mortality of seedlings can be expected (Stromberg, 1993; Mahoney and Rood 1998).

Conceptually, hydromodification would follow mechanical disturbance that prepares a seedbed in an area that either naturally receives floodwater in the spring or where bank lowering has reconnected a portion of the floodplain to the channel. Overbank flooding, and management of the receding limb of the hydrograph would then be coordinated to optimize site conditions for cottonwood and willow establishment. In sites that are not susceptible to spring overbank flows, irrigation infrastructure would be necessary to control the inundation and drainage during seedling establishment. Hydromodification projects would require a full understanding of the site's soil properties (texture, chemistry, and water relationships) as well as seasonal groundwater levels and flooding potential. This technique also assumes that recruitment of new willows and cottonwoods is not limited by seed availability, but that the hydrology of the site is unfavorable. As such, this technique could be employed following exotic species removal and surface disturbance.

Cottonwood seedlings have been shown to survive groundwater decline rates of 2 to 4 cm/day (Mahoney and Rood, 1991, 1998; Taylor et al., 1999). At Bosque del Apache, Sprenger et al. (2002) demonstrated that staged drawdowns of 5 cm/day were too fast and cottonwood seedlings died due to excessive moisture stress. Mechanical saltcedar removal resulted in higher seedling densities of cottonwood, willow and saltcedar compared to chemical control (Sprenger et al., 2002). Horton and Clark (2001) showed an inverse relationship between water table decline rates and the growth and survival of potted Goodding's willow under controlled drawdown. They also demonstrated greater root elongation rates in saltcedar in response to water table declines, resulting in higher survival rates compared to willow. Root elongation rates for Goodding's willow are slower than both saltcedar (1.3-3.8 cm/day; Horton and Clark, 2001) and cottonwood. Taylor et al. (1999) reported that when overbank sites were first cleared of vegetation and the river stage declined at slower rates, cottonwood and Coyote willow had greater densities and higher survival rates.

Overbank floods that receded in late June appeared to create higher densities of saltcedar seedlings compared to cottonwood (Taylor et al., 1999). Such results stress the importance of appropriately timed flow and hydrological manipulations to achieve native plant regeneration. The practicable application of flow and drainage manipulation to encourage native species regeneration will differ within each reach and will need to be determined using flow modeling and other tools. Moreover, irrigation infrastructure may be necessary at upland sites to control drawdown sufficiently to match the fairly tight rates of root elongation.

In general, the success of hydromodification projects for recruiting native vegetation depends on two well-timed events that must occur in tandem: (1) a late May-early June flood for native seed germination, and (2) a gradual stage decline that corresponds to root elongation rates. Overbank events after mid to late June might better prepare the site for saltcedar germination than for native species. Also, short spring pulses may not provide sufficient water to willows and cottonwoods as their roots elongate. Moreover, the rate of stage decline (natural or dam-controlled) is a function of the thickness of the capillary fringe, soil texture, floodplain elevation, and groundwater levels. Assuming that the roots of cottonwood seedlings can extend 72 to 162 cm in the first growing season (Fenner et al., 1984; Mahoney and Rood, 1998) and that a soil-water decline rate of no greater than 2.5 cm/day is needed to support the seedlings (Mahoney and Rood, 1998), a gradual stage-controlled decline in the water table would need to extend for 29 to 65 days. Whether the current river hydrology or administration could produce such flows is uncertain. Thus, restoration projects that rely on natural overbank flows and stage declines may not meet the specific hydrologic requirements for establishing riparian communities dominated by native plants.

4.5.3.6 Wetlands

The creation of wetlands could provide additional riparian habitat diversity. Wetland creation may involve ground surface lowering on terraces to create a zone with a high water table or the diversion of surface water to form an area that is seasonally saturated. Where the soil surface is lowered (i.e., depressional wetlands) capillary rise from the phreatic surface results in soil-water conditions that are amenable to riparian plants with high-water-use requirements.

Purpose of Technique. The purpose of this technique is to provide a near-surface soil-water regime that is wet enough to support wetland vegetation communities. Willows, rush-cattail-sedge, and saltgrass meadows are typically associated with shallow water tables. These plant communities not only increase overall vegetation diversity in the MRG, but may also enhance flycatcher habitat.

Considerations. Both depressional and surface water wetlands constructed for flycatcher habitat would likely focus on sites that are no longer inundated by flooding, yet have the potential to recruit breeding pairs. Specifically, sites appropriate to support flycatcher habitat need a water table sufficiently near the surface to support dense willow growth.

For depressional wetlands, the depth to the water table and soil texture will determine the height of the capillary rise and the resultant plant community. Evapoconcentration of salts will ultimately result in the development of saline soil conditions that may limit the growth of some plants. Selective planting strategies like interseeding and transplanting could also be considered in some areas to enhance the plant species diversity without impacting established plant communities.

Habitat Implications. The establishment of wetland vegetation and associated dense, shrub communities are expected to improve overall riparian community diversity and flycatcher habitat.

4.6 Depletions and Habitat Restoration

Methods to quantify depletions associated with restoration activities are not fully developed for the MRG (Section 2). Nonetheless, a qualitative assessment of the relationships between depletions and aquatic habitat restoration practices is provided below.

Most active aquatic restoration practices described herein are designed to increase the heterogeneity of flow velocities and provide more low-velocity habitats for various life stages of the minnow. In general, this would be accomplished in one of two ways: (1) lowering the bank surface to create more opportunities for overbank flows, or (2) increasing the overall length of the river. The likely consequence of these activities includes a relative rise of the water table, a decrease in flow transmission efficiency, and an increase in the free water surface area. Such results may be permanent, as with river bar and island enhancement that eventually may lengthen the channel. Other practices, such as high-flow ephemeral side channels, would only carry water temporarily. Nonetheless, most restoration activities are expected to increase basin depletions.

It is difficult to generalize the impacts on depletions for certain activities. Channel widening will increase the transmission efficiency at high flows, but may increase evaporation at low flows if the area of wetted channel increases. Reconnecting arroyos could reduce depletions locally if water is not impounded and the associated riparian plant communities die. The introduction of woody debris is not expected to significantly affect depletions. Island and bar destabilization is expected to decrease depletions by increasing flow transmission efficiency, removing vegetation, and potentially lowering the water table.

Short-term reductions in basin depletions may occur following vegetation removal, but this effect is expected to be transient as the site is revegetated. Water salvage projects that involve the conversion of phreatophyte plant communities to xeric shrub/grasslands may reduce depletions over the long-term. However, these sites may require active maintenance to prevent the reestablishment of phreatophytes.

4.7 Costs Associated with Habitat Restoration Projects

Direct costs associated with most active aquatic restoration projects will include survey and design engineering, vegetation and jetty-jack removal, and spoil hauling and disposal. Costs will also depend on a number of site-specific factors (e.g., topography, infrastructure, access) and are difficult to predict in a general sense. Design requirements for large projects, like gradient control structures, may be significant given the potential for upstream and downstream effects. Detailed hydrologic analyses may be necessary for such projects or for those in more highly populated areas. Construction costs will also vary depending on the selected design, materials used, and topographic factors.

Estimated costs for particular activities are continually being refined as work in the MRG proceeds. Costs for jetty jack removal depend on accessibility, location (in bank or in river), extent that the jack is buried, and the water content of the soil around the jack, and can range from \$140 to \$200 per jack. Installation costs for fixed logs have been estimated at \$2,000 (T. Wesche, Hab. Tech, pers. comm.). Excavation costs may range from \$8,000 to \$10,000 per acre-foot of material removed, though this estimate may be highly dependent on hauling distance and spoil disposal. Costs associated with the Los Lunas Habitat Restoration Project indicated that project costs, including design, construction, revegetation, monitoring, and so on, can approach \$20,000 per acre (Mark Horner, USACE, Albuquerque District Office, pers. comm.).

All restoration projects will likely require some level of baseline biological survey and post-treatment monitoring. These would be necessary to compare pre- and post-project effects on the minnow and/or the flycatcher. Additional monitoring may be needed to ensure that the installations function properly.

Long-term maintenance could add significant costs depending on the performance objectives and design for any particular project. For example, woody debris placements may require monitoring to ensure that materials don't become hazards. If woody material becomes dislodged, it may be necessary to stabilize or remove it to protect existing infrastructure. To extend the life expectancy of bank-line embayments or high-flow side channels, sediment may need to be periodically removed to maintain their performance. Long-term monitoring will be necessary when removing lateral confinements to ensure public safety and protect riverside facilities. If the river begins to threaten public safety or property, additional expenditures may be needed to correct the problem.

For passive restoration activities, costs are difficult to predict because they depend on site-specific factors like the extent of jetty jacks and the inherent durability of levees. Costs are also dependent on the level of intervention used to initiate island development or bank erosion. Despite the absence or minimal use of engineered structures, passive restoration designs may require detailed engineering, hydrologic, and geomorphic analyses to predict planform and cross-section changes and to ensure public safety.

5.0 PRIORITIES AND IMPLEMENTATION OF HABITAT RESTORATION PROJECTS

The Program's habitat restoration activities are currently implemented through a competitive solicitation process, whereby the Subcommittee develops an annual request for proposals (RFP) and then reviews proposals submitted by various public, private, non-governmental, and tribal entities. Each proposal is ranked according to criteria set out in the RFP. The ranked proposals are then forwarded to the Program Management committee for review to ensure the highly ranked projects meet the goals and objectives of the Program. The Program Steering Committee is ultimately responsible for implementation of Program restoration activities and makes the final determination on the award of funding.

The Program has established that actions it implements will be ranked based on three priority levels:

Priority 1: Actions that prevent extinction in the wild/do not diminish the likelihood of survival and recovery.
Priority 2: Actions that can be implemented in the near term to either test restoration concepts, or provide near-term benefits to the listed species.
Priority 3: Actions that require more comprehensive research, planning, or design to benefit the listed species or to resolve scientific or economic issues.

Actions associated with all three-priority levels may proceed simultaneously depending upon the availability of funding. Priority 1 actions to prevent extinction should receive the highest funding priority. The Priority 1 actions may be implemented in a single year or repeatedly over a number of years. Priority 2 and 3 actions will depend upon the availability of funding and the need to balance intermediate and long-term actions to provide benefits to the listed species. Following these priorities in the selection of habitat restoration projects should lead to the implementation of the most effective actions to enhance habitat, increase populations, and contribute to the recovery of listed species in compliance with applicable laws while minimizing impacts on water management, operations, maintenance, and water users and Rio Grande Compact deliveries. Habitat restoration should be planned and designed to complement other species recovery efforts such as artificial propagation and population augmentation. The remainder of this section outlines broad Program priorities that serve to guide the development of RFP's, RFP ranking criteria, river reach specific restoration plans, and specific habitat restoration projects.

5.1 Habitat Requirements and Restoration Priorities

The biology, ecology, and habitat relationships of the silvery minnow and flycatcher were discussed in Section 3.0. Section 5.1.1 summarizes key components of habitat for the silvery minnow, while Section 5.1.2 summarizes flycatcher habitat needs. The silvery minnow and flycatcher are both components of the Rio Grande ecosystem. However, the reproductive biology of the silvery minnow requires a broader restoration strategy that involves planning for contiguous habitat over substantial areas, whereas, flycatcher habitat restoration may be accomplished through activities conducted at discrete and unrelated locations. Section 5.1.3 lists priorities activities and locations for silvery minnow and flycatcher habitat restoration in the MRG.

5.1.1 Silvery Minnow Habitat Requirements

The lack of egg and larval retention habitat and river fragmentation by diversions has been identified as primary factors limiting recruitment of the silvery minnow. The amount of habitat for egg and larvae retention and young-of-year must be increased significantly to prevent the extinction of the fish in the wild. For the population to be self-sustaining in the MRG, it must be established in one or more reaches with consistent water supplies. Furthermore, augmentation (stocking) will be required initially to help establish viable populations in the upstream reaches, until the population stabilizes. Habitat restoration efforts for the silvery minnow will require concerted efforts that involve water management, habitat restoration, and population management. The suite of actions necessary for habitat restoration to meet the goal of egg and larvae retention and young-of–year rearing habitat are summarized below.

A) Sustained flows in key reaches to promote sufficient populations of wild silvery minnows

The first and fundamental issue regarding the conservation and recovery of any fish in arid environments is that fish need water to survive. The 2003 BO defines water delivery requirements to benefit silvery minnows in the MRG during wet, moderate and dry water years (Section 2.4.5). Because channel drying occurs in the Isleta and San Acacia reaches under the current hydrologic regime, silvery minnow recovery efforts should be focused on increasing populations in the reaches between Cochiti Reservoir and Isleta. These reaches are expected to have more consistent flows in the foreseeable future. The clear waters immediately below Cochiti Dam may aggravate local predatory pressures on the silvery minnow and should be considered in aquatic restoration efforts upstream of Angostura.

B) Spring flow peak in mid- to late-May to stimulate spawning

As discussed in Section 3.1.3.3, silvery minnows can spawn from April through at least June, with peak egg production occurring in mid- to late-May. Conditions that trigger spawning are not completely understood, but peak spawns are generally correlated with increased flows and turbidity levels in the spring. Establishing viable wild populations of silvery minnow will require the release of a spring flow spike, if a natural flow spike does not occur, to stimulate spawning. The size of the flow spike does not necessarily have to be large (i.e., 1,000 to 3,000 cfs or greater) to induce spawning. Available data indicate that flow increases of several hundred cfs have induced spawns. Larger peak flows should be considered with respect to their positive benefits on aquatic habitat.

C) Establishment of channel conditions that retard downstream displacement of eggs and larvae

As discussed in Section 3.1.3.4, pelagic spawning results in the downstream drift of eggs and larvae for at least the first 3 to 5 days following spawning, until the larvae develop sufficiently to escape the flow. Maintenance of viable upstream populations requires a sufficient density of habitat features that reduce the rate and magnitude of downstream transport of eggs and larvae. Therefore, the success of the restoration practices aimed at egg and larval retention will depend on maximizing the upstream capture and retention of eggs and larvae. This may be accomplished by managing river flows during the spawn as described above and by restoring the diversity of habitat features that promote shear zones with net-zero flow velocities. Restoration activities implemented to retain eggs and larvae should be coordinated with upstream population augmentation efforts. Then, as the density of spawning silvery minnows increase, the overall benefits accrued through restoration activities should dramatically increase reducing the need for augmentation.

D) Establishment of a sustainable population of silvery minnows in the Angostura reach

Section 3.1.2.2 described how populations of silvery minnows in the reach between Otowi to Bernalillo may have historically served as a "brood stock" that maintained downstream populations. Water supply is fairly consistent in between Cochiti Dam and Isleta diversions (relative to the lower reaches) and a sustainable population in this area would supply eggs and larvae to downstream reaches. Because the existing population density of silvery minnows in these upstream reaches is low, initial reestablishment of silvery minnows would depend on supplemental stocking from the refugia, hatchery, and laboratory brood and culture stocks. In an effort to maximize the genetic diversity of the upstream stock, while also maintaining the diversity of the entire downstream population, transplants of the individuals from downstream to upstream stocks may be needed.

E) Establishment of suitable feeding and cover habitat for juveniles and adults

As discussed in Section 3.1.4.2, juvenile and adult silvery minnows are typically captured in lower velocity zones, downstream of debris piles, and along shorelines. Increasing the availability of low velocity zones in the channel and deeper water scour pools would provide both summer and winter habitat.

F) Remediate longitudinal discontinuity associated with irrigation diversion structures

As discussed in Section 3.1.2.2, irrigation diversions and Cochiti Dam have fragmented the longitudinal continuity of the river affecting silvery minnow movements to upstream reaches. The construction of fish passage structures at the irrigation diversions may reduce and potentially eliminate concerns associated with habitat fragmentation for the silvery minnow.

5.1.2 Flycatcher Habitat Requirements

Flycatcher breeding and migration habitats occur in the MRG. The priorities for flycatcher habitat restoration are problematic, since significant areas of suitable habitat exist in the MRG, but are not occupied (Moore and Alhers, 2003). The apparent surplus of habitat suggests that factors other than habitat availability may limit recruitment (Section 3.2.4.4). Furthermore, it is likely that additional habitat will develop just south of the Program Area if the water levels in Elephant Butte reservoir continue to decline.

As discussed in Section 3.2.4, flycatchers are most commonly found in vegetation stands or patches that are 0.6 ha (1.5 ac) or larger. Breeding territories consist of dense vegetation, or dense patches interspersed with openings containing water or shorter stature/sparser vegetation. Thickets of trees and shrubs used for nesting range in height from about 2 to 30 meters (6 to 100 feet), with dense vegetation occurring mostly within the first 3 to 4 meters (10-13 feet) above the ground. Cottonwood gallery forests that lack a dense understory of shrubs and trees do not provide breeding habitat for flycatchers. Flycatcher habitat restoration efforts can be effectively implemented in discrete locations, although it is unclear how to attract breeding pairs to restored sites.

5.1.3 Restoration Priorities

Although both species are endangered, the silvery minnow is currently at much greater risk of extinction along the MRG than the flycatcher. Consequently, measures to prevent the extinction of the minnow are urgently needed. Some of these measures fall within the category of habitat restoration, whereas others involve flow maintenance, captive propagation, and minnow salvage efforts that are Program concerns but outside the scope of the current habitat restoration plan. For the next three to five years, the emphasis will be placed on silvery minnow augmentation and habitat restoration efforts. During this time period limited funding for protecting existing flycatcher habitat and preparing sites for flycatcher habitat restoration may be considered. Thereafter, the Subcommittee anticipates that emphasis can then appropriately shift to improving flycatcher habitat and general ecosystem restoration. This priority approach is not meant to diminish the importance of maintaining quality flycatcher habitat or general riparian restoration, but rather emphasizes the need to allocate limited resources to resolve the most pressing issues associated with the silvery minnow. The following prioritized activities (listed in order of importance) are those that the Subcommittee believes best address the limiting factors described above. Consultation with the Pueblos and tribes would take place to encourage planning and implementation opportunities on their lands.

Silvery Minnow Priorities

1) Plan, design, and implement aquatic habitat restoration for the silvery minnow, emphasizing egg and larval retention features and refugia, in the reach from Cochiti Dam to Isleta Diversion, where perennial flows in the MRG are most reliable. Restoration techniques may include, but are not limited to, techniques described in Section 4.
2) Plan, design, and implement aquatic habitat restoration for the silvery minnow, emphasizing egg and larval retention features and refugia in the reach from Isleta Diversion to San Acacia Diversion, where perennial flows in the MRG are somewhat less reliable than in the upper reaches. Restoration techniques may include, but are not limited to, techniques described in Section 4.
3) Plan, design and construct fish passages concurrently at Angostura and San Acacia diversions, followed by comparable efforts at the Isleta Diversion.
4) Plan, design, and implement aquatic habitat restoration for the silvery minnow, emphasizing egg and larval retention features and refugia, in the reach below San Acacia Diversion, where perennial flows in the MRG are least reliable. Restoration techniques may include, but are not limited to, techniques described in Section 4.

Flycatcher Priorities

1) Protect and maintain existing flycatcher territories in the Program Area (i.e., at Espanola/San Juan Pueblo, San Marcial, Sevilleta NWR) by planning, designing and implementing projects to reduce fire danger, riparian vegetation loss, and exotic vegetation encroachment, and to maintain vegetation structure required by this species.
2) Plan and design exotic vegetation removal projects in other areas of the MRG to take advantage of anticipated overbank flooding opportunities to create flycatcher habitat. The subcommittee expects that site preparation activities that emphasize passive revegetation (Section 4) will be most cost efficient for maximizing benefits associated with the reestablishment of native plants through spring flooding.
3) Plan, design, and implement habitat restoration for flycatcher in riparian areas adjacent to existing flycatcher nesting areas within the Program Area (i.e., Espanola/San Juan Pueblo, San Marcial, and Sevilleta NWR).

5.2 Habitat Restoration Evaluation Criteria

The Subcommittee evaluates the habitat restoration proposals based on their relative merits. The primary evaluation criteria are briefly described in the following sections. In general, projects that incorporate passive restoration strategies are likely to provide superior long-term performance relative to purely active restoration techniques that will require maintenance.

5.2.1 Benefits to Species of Concern

Primary consideration is given to habitat restoration projects that directly benefit the silvery minnow and/or flycatcher. The projects should provide sustainable habitat for a particular life stage or succession of life stages of the silvery minnow. This criterion deals primarily with technical feasibility issues related to the biology and ecology of the silvery minnow and flycatcher, but may also be evaluated with respect to geomorphic and hydrologic considerations.

5.2.2 Feasibility of Implementation

Feasibility of implementation is an important consideration for habitat restoration projects. Land access, environmental compliance (e.g., National Environmental Policy Act, Endangered Species Act, Clean Water Act), Federal, State, and Municipal permits, and administrative feasibility must be addressed.

Construction activities must not be conducted during flycatcher breeding season in or near flycatcher habitat. Restoration projects will be evaluated comprehensively to include issues of ingress, egress, staging, storage, as well as, on-site construction. Feasibility will also be evaluated from a long-term perspective including schedules and budgets for multi-year projects and provisions for maintenance and monitoring.

5.2.3 Water Requirements and Depletions

Because water in the Middle Rio Grande basin is fully appropriated and a large portion of the river flow must be delivered to Texas under the Compact, changes in the amount of water needed for habitat restoration or to achieve mandated target flows must be considered by the Program. Habitat restoration projects will be evaluated with respect to net depletions and specific flow requirements (Section 2.7). Projects that result in increased depletions must account for mechanisms to offset depletions within the basin. Ultimately, if additional water is needed for endangered species habitat restoration an existing water use must be suspended.

5.2.4 Sustainability and Maintenance

The sustainability of restored habitat and the need for maintenance are important considerations for evaluating projects. Habitat features that are self-sustaining offer the greatest potential for long-term benefits. In this respect, the projects should be developed in a manner consistent with the hydrologic and geomorphic regime of the river including the potential consequences of extreme events. For structures or projects that require maintenance, consideration should be given to the life expectancy, frequency of maintenance, contingencies for extreme events, and responsibility for long-term maintenance.

5.2.5 2003 Biological Opinion

The Program recognizes the need to address components of the BO. Projects will be evaluated with respect to their ability to achieve the goals and objectives of the BO. In particular, the BO identified goals for restoration activities in eight reaches below Velarde, indicating, “By 2013, additional restoration totaling 1,600 acres will be completed in the action area… The action agencies and parties to the consultation…..shall develop timetables and prioritize areas for restoration. Projects should result in restoration/creation of blocks of habitat 60 acres or larger.”

In recognition of the drought-related water supply issues currently facing the MRG, the BO identified geographic priorities in reaches with more consistent water supplies. More specifically, the BO states, “in the short-term (five years or less), the emphasis for silvery minnow habitat restoration projects shall be placed on the river reaches north of San Acacia.” Furthermore, Senator Domenici indicated that in the absence of supplemental water, habitat restoration should be conducted where there is sufficient water to maintain the silvery minnow. This is a sensible approach based on a review of past conditions and stochastic water supply predictions. During the 1950’s drought the river dried below Albuquerque for sustained periods of time (as much as 100 days per year in some years). Given that we have entered what may be another extended drought and that Article VII of the Compact is in effect, which limits upstream storage, focusing on aquatic habitat restoration work upstream from Isleta Diversion seems warranted in the short term. Beyond considerations of water reliability, emphasis on upstream aquatic restoration is prudent from an ecological perspective, because it will enhance a sustainable brood stock. Restoration activities in the upper reaches should focus on the development of egg and larval retention and nursery habitat during spring peak flows.

5.2.6 Costs

Because the restoration program is based on a competitive process the costs for the proposed project will be evaluated relative to the benefits to the listed species. Cost is used here to refer to direct and indirect costs associated with design, construction, maintenance, and monitoring that would be incurred over the life of the project. Benefits for the listed species and water related costs, as discussed above, would be compared to the total costs on a project-by-project basis to optimize the value to the Program.

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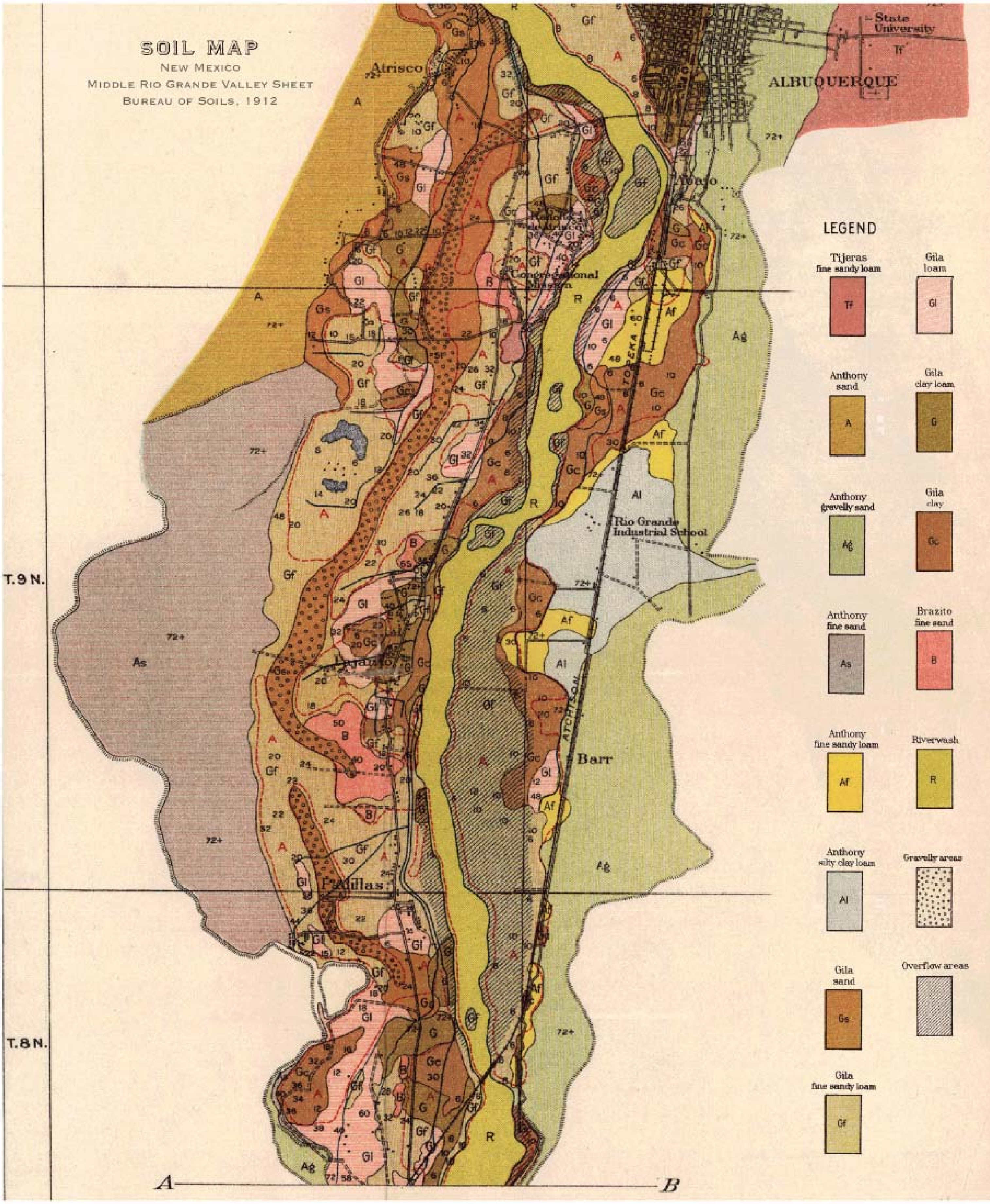
SOIL MAP
NEW MEXICO
MIDDLE RIO GRANDE VALLEY SHEET
BUREAU OF SOILS, 1912
Atrisco
ALBUQUERQUE
State University
Congregational Mission
Rio Grande Industrial School
Barr
Padillas
T.9 N.
T.8 N.
TOPEKA
ATCHISON
LEGEND
Tijeras fine sandy loam
Gila loam
Anthony sand
Gila clay loam
Anthony gravelly sand
Gila clay
Anthony fine sand
Brazito fine sand
Anthony fine sandy loam
Riverwash
Anthony silty clay loam
Gravelly areas
Gila sand
Overflow areas
Gila fine sandy loam