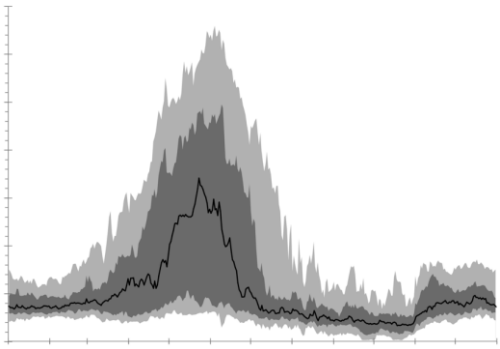
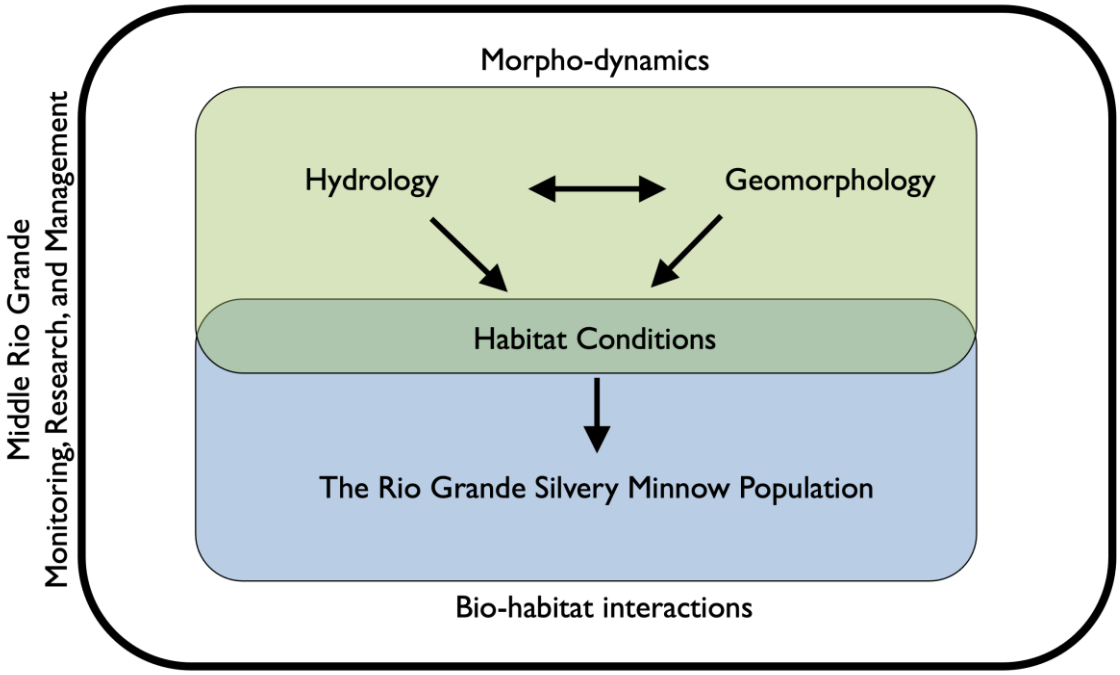
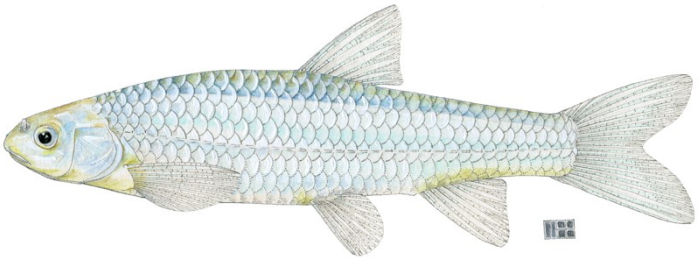
Linking Morpho-Dynamics and Bio-Habitat Conditions on the Middle Rio Grande: Process-Linkage Report III – Angostura Reach Analyses

Final Report

A U.S. Bureau of Reclamation Funded Research Project







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15 December 2023

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EXECUTIVE SUMMARY

This report (the Process-Linkage Report) is the last in a series of reach-scale reports (I–Isleta, II–San Acacia, and III–Angostura) produced for a collaborative research project initiated by the U.S. Department of the Interior – Bureau of Reclamation, in collaboration with Colorado State University – Department of Civil and Environmental Engineering, University of New Mexico – Department of Biology, and American Southwest Ichthyological Researchers, L.L.C.

The principal objectives of this study of the Middle Rio Grande, NM were to:

- Identify and assess linkages between the observed morpho-dynamics and biological habitat conditions
- Improve understanding of the specific morpho-dynamic processes that are hypothesized to influence the habitat conditions and population dynamics of the Rio Grande Silvery Minnow
- Provide recommendations for data collection to fill in observed data gaps in the characterization and assessment of process-linkages
- Provide recommendations for river management practices that have potential to create and maintain suitable habitat for the Rio Grande Silvery Minnow

This study performed interdisciplinary analyses to improve understanding of the linkages among dynamic hydrologic and geomorphic processes (i.e., morpho-dynamics) and the hydraulic habitat conditions needed by the Rio Grande Silvery Minnow, a federally endangered species. We used a suite of analytical methods to integrate several long-term, systematically collected datasets that were designed to monitor and characterize hydrologic, geomorphic, and ecological trends in the Middle Rio Grande. This study furthered efforts to understand relationships between hydrogeomorphic processes and ecological dynamics occurring at the reach-scale (i.e., the Angostura Reach). We characterized relationships between discharge and habitat availability (temporally and spatially), developed a habitat metric incorporating hydrologic, geomorphic, and ecological factors over time, evaluated long-term ecological relationships between the Rio Grande Silvery Minnow and environmental conditions, and described key linkages among morpho-dynamic processes and habitats needed by the Rio Grande Silvery Minnow and their potential management implications.

Building on prior Process-Linkage Reports, the main findings for the Angostura Reach included:

- Key process-linkages identified for the Middle Rio Grande were: (1) floodplain connectivity and inundation, (2) hydrologic connectivity (within and among reaches), and (3) main channel habitat complexity and availability.
- Hydrologic and geomorphic conditions within the Angostura Reach showed spatially uniform trends over time (1962–2012) that were similar to those observed in the Isleta Reach (Process-Linkage Report I).
- Discharge was consistently higher in the Angostura Reach relative to downstream reaches, however, differences in TIHMs across reaches varied over time (1993–2021).
- Larval TIHMs for the Angostura Reach showed higher contributions from the Bernalillo subreach 1992–2012); habitat maps showed secondary channels created hydraulically suitable habitat when inundated.
- Densities of the Rio Grande Silvery Minnow were generally lower in the Angostura Reach than the San Acacia Reach but varied relative to the Isleta Reach. Differences in densities across reaches were attributed to differences in larval habitat availability and ecological processes (i.e., drift/dispersal).
- TIHMs and flow metrics corresponding to the larval life-stage of Rio Grande Silvery Minnow (May–June) were the most reliable long-term predictors of the species' density and occurrence across reaches, however, flow metrics explained more variation in population parameters across years.
- Data gaps and analytical considerations were identified – principally, collection of channel and floodplain elevations across flows, particularly low flows, is needed to improve modeling accuracy across reaches. Limitations to hydraulic models and habitat analyses included reduced modeling accuracy at low flows and uncertainty associated with overbanking thresholds for various channel states (e.g., incised [Angostura] vs. perched channels [San Acacia]). Targeted 2D modeling was recommended.
- Flow management and habitat restoration in the Angostura Reach will be important to the recovery of the Rio Grande Silvery Minnow in the Middle Rio Grande. The upstream location of this reach is important for providing spawning and nursery habitats to counter net downstream displacement of offspring. Low floodplain connectivity suggests important opportunities to increase larval habitat availability through construction of floodplains and low-velocity side channels (i.e., increase main channel complexity). Higher flow magnitudes in the reach also hold potential for increasing larval habitat during spring runoff and for improving survivorship during low flow periods. However, incised channel morphology and prevalence of infrastructure, municipal land use, and bank armoring features (i.e., jetty jacks) limit potential for recovery via natural processes.

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INTRODUCTION

Purpose and Objectives

This report (the Process-Linkage Report) is one product of a collaborative research project initiated by the U.S. Department of the Interior – Bureau of Reclamation, in collaboration with Colorado State University – Department of Civil and Environmental Engineering, University of New Mexico – Department of Biology, and American Southwest Ichthyological Researchers, LLC. The objectives of the Process-Linkage Report are to:

- Identify and assess process-linkages between the observed morpho-dynamics and biological habitat conditions in the Middle Rio Grande, NM
- Improve understanding of the specific morpho-dynamic processes that are hypothesized to influence the habitat conditions and population dynamics of the Rio Grande Silvery Minnow
- Provide recommendations for data collection to fill in observed data gaps in the characterization and assessment of process-linkages in the Middle Rio Grande, NM
- Provide recommendations for river management practices that have potential to create and maintain suitable habitat for the Rio Grande Silvery Minnow

This is the third and final Process-Linkage Report produced for this project, which includes an assessment of process-linkages for the Angostura Reach of the Middle Rio Grande (Bernalillo, NM to Isleta Diversion Dam) – the preceding Process-Linkage Reports targeted the Isleta and San Acacia Reaches (Mortensen et al., 2020; 2023). Insights gained from the Isleta and San Acacia Reaches included:

- Key process-linkages identified were (1) floodplain connectivity and inundation, (2) hydrologic connectivity within and among reaches, and (3) main channel habitat complexity and availability,
- Isleta Reach geomorphology was generally characterized by channel narrowing and incision over time caused by reduced sediment supply, channelization, and encroachment of nonnative riparian vegetation; San Acacia Reach included channel aggradation downstream of Escondida, NM,
- Geomorphic trends generally showed negative impacts to process-linkages related to increased bankfull discharge (reduced floodplain connectivity) and reduced availability of shallow, low-velocity habitats in the main channel,
- Analyses of long-term ecological relationships indicated that flow metrics (as compared to habitat metrics [TIHMs]) consistently explained the most variation in the Rio Grande Silvery Minnow population over time (1993–2021).

This Process-Linkage Report presents the final reach analyses completed for this study. This report reflects the contributions of numerous collaborators over several years to refine the methods, analyses, and results during the planning, preparation, and completion of previous reporting.

Project Background and Motivation

The Middle Rio Grande has experienced significant geomorphic changes resulting from river engineering activities (e.g., channelization, dam construction), reduced peak flows during spring runoff (i.e., reduced magnitude, duration, and frequency), increased duration and frequency of low flows, establishment of riparian vegetation, and complex sediment dynamics (e.g., channel incision, plug formation, coarsening of the riverbed, and variable tributary inputs). Coincident with hydrologic and geomorphic impacts to the Middle Rio Grande, the Rio Grande Silvery Minnow *Hybognathus amarus* (RGSM) population has declined precipitously, motivating its listing as a federally endangered species (USFWS, 1994). The primary threats to this species include alteration of the natural hydrograph and habitat loss. Alterations to the hydrograph and channel morphology have synergistically decreased the availability and persistence of spawning and nursery habitats and reduced the frequency and magnitude of recruitment events. Investigation of the closely linked interactions among hydrology, geomorphology, and habitation conditions of this species is needed to fully understand species recovery and persistence.

The U.S. Bureau of Reclamation (USBR) holds responsibility for maintaining the river channel through the Middle Rio Grande (Flood Control Act of 1950). Accordingly, USBR plays an active role in research and monitoring efforts designed to inform management of flows, aquatic habitats, and ecological resources. This study is pursued under the supervision of USBR to investigate the complex dynamics of habitat conditions needed by the Rio Grande Silvery Minnow. This project is a collaborative, interdisciplinary study of the Middle Rio Grande ecosystem by researchers at the Colorado State University (Fort Collins, CO), the University of New Mexico (Albuquerque, NM), and American Southwest Ichthyological Researchers (ASIR, Albuquerque, NM).

In the past two decades, the Colorado State University Department of Civil and Environmental Engineering (CSU) has completed numerous studies on the fluvial geomorphology of the Middle Rio Grande including, hydrology and hydraulics, bed material and sediment transport, bed forms, changes in planform and channel geometry, and sediment plug formation. Several reports produced by CSU for USBR have documented past and present (ca. 1918–2020) geomorphic changes and processes. Generally, these changes have increased the homogeneity of the Rio Grande and reduced the availability and complexity of habitats across the current range of expected flows.

Over the past two decades, the University of New Mexico (UNM) and American Southwest Ichthyological Researchers, LLC (ASIR; jointly UNM-ASIR) have studied and systematically monitored the biology, population dynamics, and habitat conditions of the Rio Grande Silvery Minnow (e.g., Dudley and Platania, 1997; Osborne et al., 2006; Pease et al., 2006; Carson et al., 2020; Dudley et al., 2022). Research on the Rio Grande Silvery Minnow has shown a propensity for specific habitats during periods related to its life history and reproductive strategy. These habitats are primarily in the main river channel, but during spring runoff, overbank flows create habitats that are crucial to the spawning and recruitment of the Rio Grande Silvery Minnow. During this time, suitable nursery habitats must be available and persist for larval fish to grow large enough (i.e., juvenile life-stage) to survive in the main channel when flows recede. This ecologically significant process (i.e., spring runoff, floodplain inundation) has been reduced in magnitude and frequency by the closely linked and interacting effects of changes to hydrologic and geomorphic processes. Furthermore, the availability and complexity of main channel habitats have decreased, and the frequency and duration of low flow periods have increased, which affects the survival of the species.

There is a strong need to understand process-linkages between the studied morpho-dynamics on the Middle Rio Grande and the habitat conditions needed by the Rio Grande Silvery Minnow. Considering the increasing pressure from changing climate and water scarcity, linking the fields of engineering-geomorphology and biology-ecology will improve our holistic understanding of the complex Middle Rio Grande ecosystem. The Department of Civil and Environmental Engineering at CSU offers expertise on the analysis of technical river engineering problems including the effects of a variable hydrologic conditions and sediment loading on the geomorphology of a rapidly evolving river system. The Department of Biology at UNM provides expertise on biological and ecological interactions within complex lotic and riparian environments including the analysis of biological community dynamics and biotic habitat requirements. ASIR has systematically monitored the Rio Grande Silvery Minnow population since 1993 including foundational studies of reproductive biology, spawning periodicity, and habitat use of this imperiled species. It is expected that through an interdisciplinary collaboration involving these parties and expertise at USBR, that it is possible to identify and evaluate links between morpho-dynamics and biological-habitat conditions on the Middle Rio Grande (Figure 1). By making these linkages, additional insight into data gaps and innovative river management practices are expected, which will help identify strategies to increase the complexity of the Middle Rio Grande and restore ecological integrity.

The goal of this study is to perform interdisciplinary analyses and improve understanding of the morpho-dynamics of the Middle Rio Grande regarding the habitats of the Rio Grande Silvery Minnow. These analyses consider the spatial and temporal scales of bio-habitat conditions for the Rio Grande Silvery Minnow, providing an assessment of how long that habitat may persist, the tendency of the natural fluvial processes to continually create the desired habitat, the potential spatial scale of the created habitat, and the anthropogenic inputs that may be needed to initiate or sustain these links. The analyses incorporate multidisciplinary approaches that address floodplain connectivity, geomorphic suitability for restoration, and species needs. These investigations will seek to identify, if possible, some of the key components that determine population dynamics of the Rio Grande Silvery Minnow. Recommendations will also be made on data gaps, for which data collection may improve linking the various processes.

Recommendations may also be made, suggesting innovative river management practices that would help increase the complexity and heterogeneity of the Middle Rio Grande. This is a multi-year study jointly pursued by CSU and UNM-ASIR with feedback from USBR.

Basis for Process-Linkage Report

This section provides a brief review of interdisciplinary river research efforts and describes how previous river ecosystem studies informed the basis for this study of the Middle Rio Grande. This review identified key references from a growing body of literature and variety of sources (e.g., peer-reviewed publications, grey literature, white papers). The fundamental concepts and approaches identified in this review are described and implemented in the Process-Linkage Report. This literature review was central to the development of frameworks, models, and relationships that reflect our current understanding of the complex ecosystem dynamics occurring in the Middle Rio Grande (see process-linkage framework, conceptual models and relationships in this report).

Interdisciplinary Study of River-Floodplain Ecosystems

The study of river-floodplain ecosystems inherently involves multiple scientific disciplines, including biology, ecology, engineering, geomorphology, and hydrology. The need to understand and address ecological concerns requires comprehensive, interdisciplinary approaches that simultaneously consider the physical and biological components of river systems (Thoms and Parsons, 2002; Dollar et al., 2007; Vaughan et al., 2009; Meitzen et al., 2013; Gurnell et al., 2016; Krueger et al., 2016). The recovery of the Rio Grande Silvery Minnow presents an opportunity to improve holistic understanding and management of the Middle Rio Grande ecosystem by integrating knowledge and expertise among multiple disciplines and long-term monitoring efforts.

Several key concepts have emerged from research at the interface of hydrology, geomorphology, and ecology (e.g., ecohydrology, hydromorphology, ecogeomorphology, hydrogeomorphology…the permutations are numerous). Across these disciplines, river systems are commonly represented conceptually by a hierarchy (i.e., a graded, organizational structure or framework) of processes and features that function dynamically across multiple spatial scales over time (Frissell et al., 1986; Fausch et al., 2002; Thoms and Parsons, 2002; Dollar et al., 2007; Stillwater Sciences, 2007; Trinity River Restoration Program, 2009; Beechie et al., 2010; Meitzen et al., 2013; Jacobson et al., 2014; Gurnell et al., 2016). Hierarchical frameworks help simplify and organize the complex interactions among the suite of hydrological, geomorphological, and ecological processes that occur in riparian ecosystems. The inclusion of multiple spatial scales and temporal analyses are important for recognizing and understanding the underlying drivers of geomorphic change, the constraints imposed on current fluvial processes, and the possible evolutionary trajectories and timelines of change under future management scenarios (Grabowski et al., 2014). For example, understanding reach scale hydromorphology requires knowledge of processes and human pressures at not only the reach scale (e.g., 1–10 km) but also at larger spatial scales including the catchment scale (e.g., 10^2 – 10^5 km^2; Gurnell et al., 2016). Additionally, numerous approaches to interdisciplinary river research emphasize process-based principles (Frissell et al., 1986; Fausch et al., 2002; Stillwater Sciences, 2007; Trinity River Restoration Program, 2009; Vaughan et al., 2009; Grabowski et al., 2014; Jacobson et al., 2014; Gurnell et al., 2016) – a focus on the normative rates and magnitudes of physical, chemical, and biological processes that create and maintain river and floodplain ecosystems (Beechie et al., 2010). Process-based approaches are suited to identify and mitigate the root causes of degradation, leading to enhanced restoration outcomes, as opposed to traditional, form-based approaches that tend to address the symptoms of morpho-dynamic alterations rather than the causes (Beechie et al., 2010; Grabowski et al., 2014). These concepts are central to numerous frameworks and approaches designed to improve interdisciplinary understanding of river ecosystems — a hierarchical structure links hydrologic, geomorphic, and ecological processes across multiple temporal and spatial scales, levels of organization, and complexity. Accordingly, these concepts were implemented in this study to the greatest degree possible given available data and methods for the Middle Rio Grande.

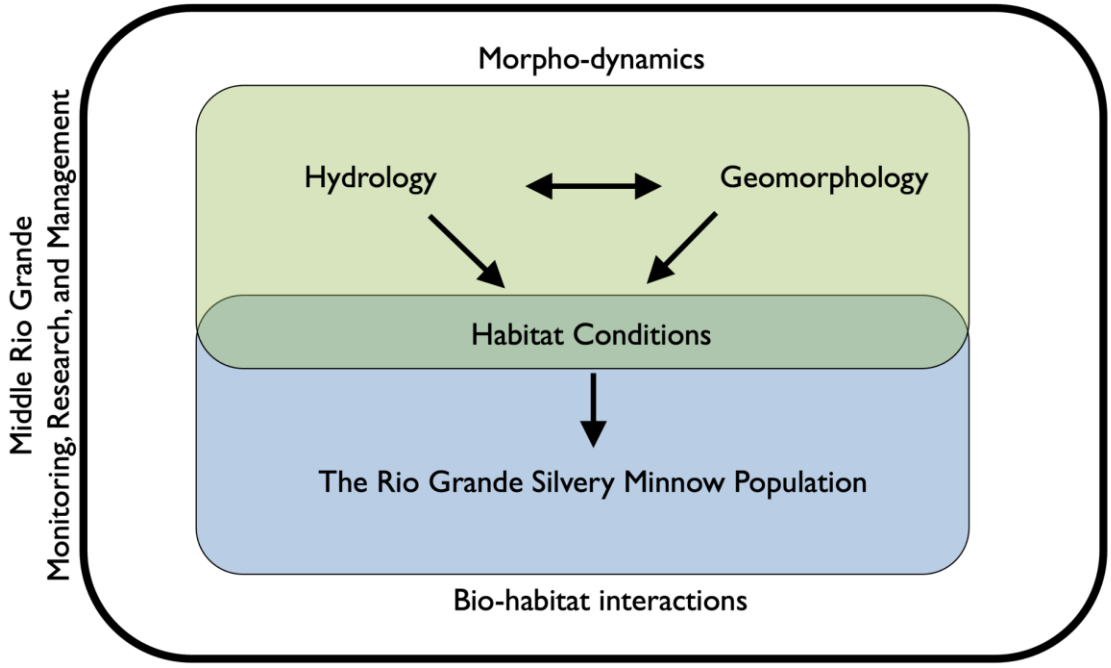


Figure 1. Simplified conceptual diagram of linkages among morpho-dynamic processes and bio-habitat interactions in the Middle Rio Grande. Boxes represent the knowledge and expertise of scientific disciplines.

Linking Morpho-dynamics and Biological-habitat conditions on the Middle Rio Grande

Previous interdisciplinary studies of large and complex river ecosystems informed the basis for linking morpho-dynamics and biological habitat conditions on the Middle Rio Grande. Notably, studies of the Sacramento River, CA (Sacramento River Ecological Flows Study), Trinity River, CA (Trinity River Restoration Program), and Missouri River basin (Missouri River Recovery Program) helped develop approaches to the Middle Rio Grande (Jacobson et al., 2014; Trinity River Restoration Program, 2009, The Nature Conservancy et al., 2008; Stillwater Sciences, 2007). Considerable efforts have begun to improve interdisciplinary understanding of fish population dynamics in other large river systems globally (e.g., Columbia River basin [USA], Murray-Darling River basin [AU]), however, not all interdisciplinary approaches are transferable to the Middle Rio Grande due to underlying regional differences in hydrology, geomorphology, or ecology resulting in study designs that are not particularly suited for the Rio Grande. Overall, interdisciplinary studies of large river systems, including this study, share common objectives such as:

- Synthesize available knowledge about fundamental ecosystem process, habitats, and native species
- Develop and refine conceptual models that illustrate key linkages between watershed inputs, fluvial processes, aquatic habitat conditions, and ecological responses
- Improve understanding of how management actions influence the creation and maintenance of habitats for native species

Interdisciplinary river research efforts provided a basis to investigate process-linkages in the Middle Rio Grande ecosystem. Specifically, several approaches and conceptual models were implemented in a conceptual framework specific to the Middle Rio Grande.

Given the complexities of large river ecosystems and the challenges of integrating scientific disciplines, researchers have also developed frameworks that provide a basic structural foundation to assess and understand ecosystem responses. Specifically, a collaborative, interdisciplinary research program, Restoring rivers FOR effective catchment Management (REFORM), recently developed a multi-scale, hierarchical framework to improve understanding of river morpho-dynamics and inform river management within the European Union (Gurnell et al., 2016). The development of the REFORM framework included the synthesis of 16 existing hierarchical frameworks and contributions from over 30 authors. This framework is among the most recent approaches to interdisciplinary river research and its primary stages provided a relatively simple, tractable approach to improve understanding of morpho-dynamic processes and responses. These stages include: (1) delineation of spatial units, (2) characterization of spatial units using existing data sets, and (3) assessment of past and present river characteristics. The three-stage REFORM approach was applied to the Middle Rio Grande, in combination with a conceptual framework, to investigate process-linkages among geomorphology, hydrology, and biological habitat conditions over time.

For this study, the terms morpho-dynamics and bio-habitat conditions were used to broadly describe the suite of physical and ecological interactions observed in the study area. As mentioned previously, terminology across interdisciplinary river studies has yet to be standardized, therefore, these terms are defined herein for clarity. Morpho-dynamics describes the linked hydrologic and fluvial geomorphic processes that determine channel and floodplain morphology through space and time. Biological-habitat (i.e., bio-habitat) conditions refers to the characteristics (e.g., timing, magnitude, and duration) of specific aquatic habitat types, such as shallow, low-velocity areas, suspected to determine ecological responses (e.g., fish density and occurrence) via biophysical interactions with focal species. Based on available datasets, we conducted several analyses to identify and evaluate key process-linkages in the Middle Rio Grande and assess their relative influence on population dynamics of a sensitive fish species.

The framework, approaches, and methods applied to the Middle Rio Grande are detailed in the process-linkage framework and conceptual models and relationships sections of this report. Refer to these sections for specific descriptions and models that illustrate how process-linkages were defined, characterized, and evaluated using available data, analytical methods, and current knowledge of the ecosystem.

STUDY AREA, HISTORICAL IMPACTS, AND FOCAL SPECIES

This section summarizes information presented in the Rio Grande Silvery Minnow Biology and Habitat Syntheses (Mortensen et al., 2019). Refer to this document for a comprehensive review of biological habitat conditions and conservation implications of hydrologic and geomorphic alteration of the Middle Rio Grande for this species.

Study Area – Middle Rio Grande

The study area is defined by the critical habitat designation for the Rio Grande Silvery Minnow under the U.S. Endangered Species Act. In 2003, the U.S. Fish and Wildlife Service (USFWS) designated the Middle Rio Grande as critical habitat for the Rio Grande Silvery Minnow (USFWS, 2003). Critical habitat defines the geographic area, and physical and biological features therein, which are essential to conserving the species. The longitudinal extent of critical habitat is defined as approximately 180 mi (290 km) of the Rio Grande downstream of Cochiti Dam. The lateral extent is defined as the area between the existing levees or by 91.4 m (300 ft) of riparian zone adjacent to each side of the bank full stage, in the absence of levees (USFWS, 2003). Critical habitat also includes the Jemez River from Jemez Canyon Dam to the upstream boundary of Santa Ana Pueblo. Lands of the Cochiti and San Felipe Pueblos are included in the designation, however, the lands of Santo Domingo, Santa Ana, Sandia, and Isleta Pueblos are excluded, and each of these Pueblos has submitted management plans that provide for special management considerations or protections for the Rio Grande Silvery Minnow.

The Middle Rio Grande (Figure 2) is divided into four reaches: (1) Cochiti Reach – Cochiti Dam to Angostura Diversion Dam (22.5 mi [36.2 km]), (2) Angostura Reach – Angostura Diversion Dam to Isleta Diversion Dam (40.8 mi [65.6 km]), (3) Isleta Reach – Isleta Diversion Dam to San Acacia Diversion Dam (53.1 mi [85.5 km]), and (4) San Acacia Reach – San Acacia Diversion Dam to the terminus of the Low Flow Conveyance Channel (57.1 mi [91.9 km] – length of this reach varies with water surface elevation at the reservoir). These diversion structures are important physical boundaries, which influence reach-scale hydrology, geomorphology, and ecology.

Process-Linkage Report III – Angostura Reach

This Process-Linkage Report focuses on an upstream segment of the Middle Rio Grande, the Angostura Reach, which is herein defined as the length of river between the U.S. Highway 550 bridge crossing at Bernalillo, NM and Isleta Diversion Dam (35.0 mi [56.1 km]; Figure 3). Due to contractually defined boundaries, the upstream extent of geomorphic data (i.e., channel geometries) terminated at U.S. Highway 550; biological data extended upstream to Angostura Diversion Dam (n= 1 site upstream of HWY 550).The Reach Reports analyzed this reach in two shorter segments (i.e., subreaches) based on previous USBR designations: (1) the “Bernalillo (B) Reach” between the U.S. Highway 550 bridge and the Montaño Bridge (Radobenko et al., 2023), (2) the “Montaño (Mo) Reach” between the Montaño Bridge and Isleta Diversion Dam (Anderson et al., 2023). Subreaches were delineated in the Reach Reports (e.g., B1–B4, Mo1–Mo5; n= 9) – these designations were maintained in this report (Figure 3, Table 1). Due to the prevalence of urban infrastructure in this reach of the Middle Rio Grande, locations of key features were primarily used to delineate subreaches. The nearest agg/deg lines were used as subreach boundaries – these lines are spaced every 500 ft, normal to the predominant flow, for systematic channel surveying over time. This reach is generally characterized by channel degradation following completion of Cochiti Dam (Figure 4; Massong et al., 2006).

Process-Linkage Report I targeted the Isleta Reach and Process-Linkage Report II targeted the San Acacia Reach. Insights and knowledge gained from Process-Linkage Reports I and II informed the preparation of Process-Linkage Report III. Consistency in analyses and formatting were maintained across reports to the extent possible, nonetheless, the exploratory nature of this study prompted the addition and modification of several items during the comment and revision periods of previous reports. Notable changes between Process-Linkage Reports are described in Data and Methods. Additionally, Process-Linkage Report III provides further comparisons of results across reaches (i.e., Angostura, Isleta, and San Acacia Reaches), however, a complete, range-wide integration of analyses and results (i.e., Middle Rio Grande) is beyond the scope of the present study.

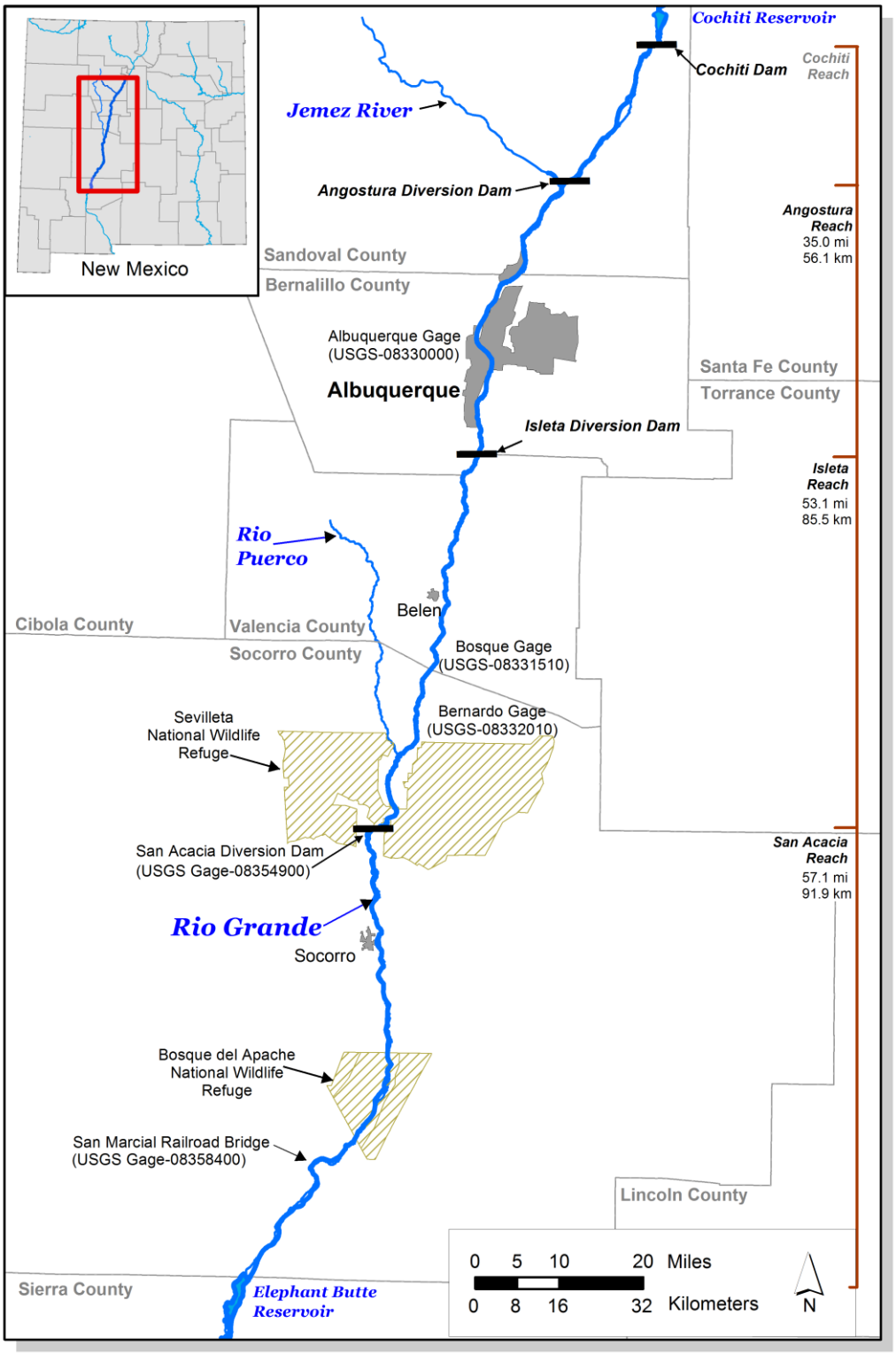


Figure 2. Map of the Middle Rio Grande, NM with selected features. Reaches between diversion dams (e.g., Angostura Reach) are shown at right.

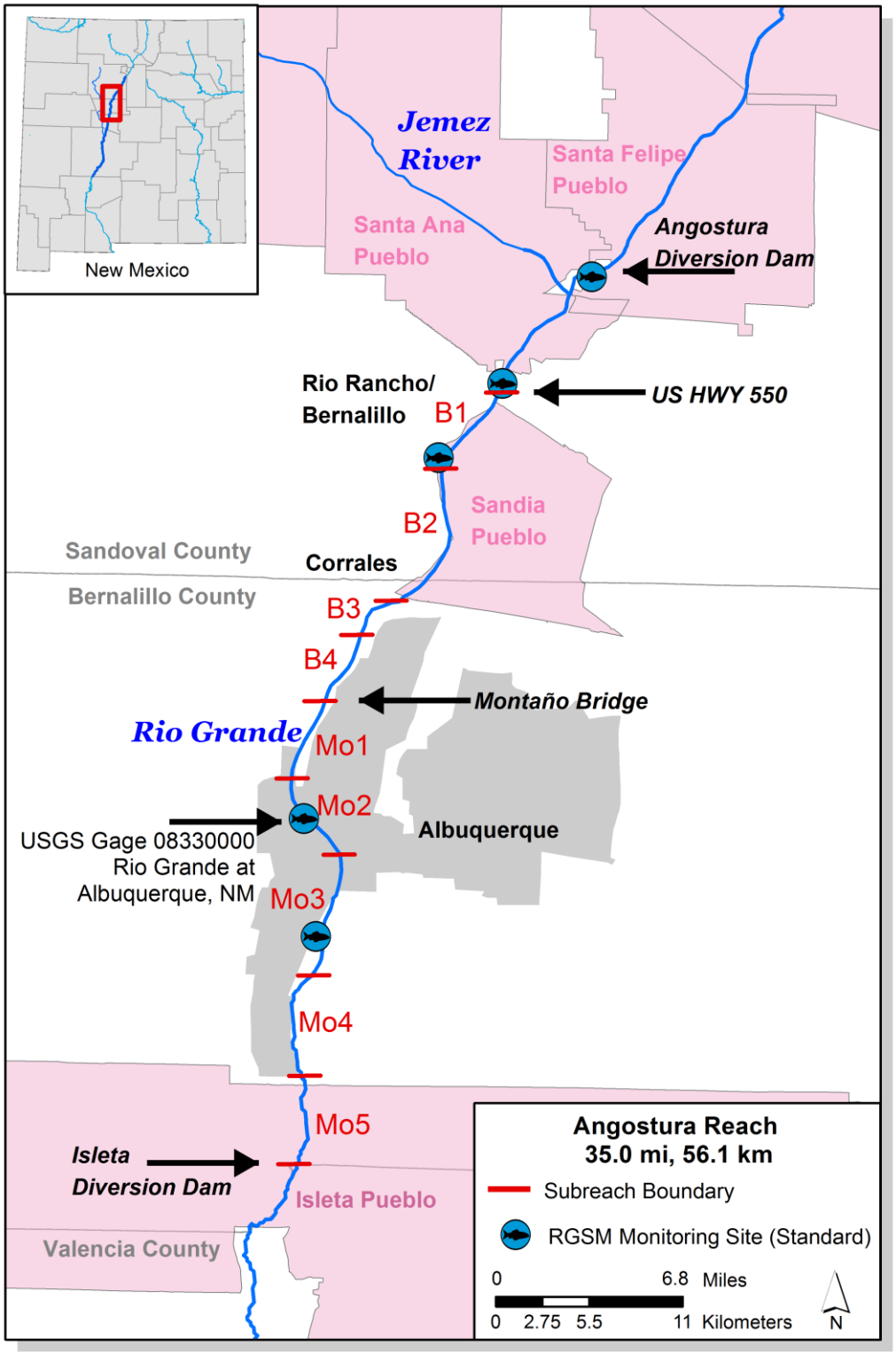


Figure 3. Map of the Angostura Reach (US 550 bridge to Isleta Diversion Dam) of the Middle Rio Grande with selected features. Gray area shows the approximate of Albuquerque, NM and pink areas show Pueblo lands.

Table 1. Subreach boundaries and characteristics for the Angostura Reach, Middle Rio Grande. Subreaches were delineated based on the geomorphic features listed.

Subreach	Agg/deg line no.	Length, mi (km)	Boundaries
B1	298–339	4,0 (6.4)	US HWY 550 to siphon crossing
B2	339–398	5.6 (9.0)	Siphon crossing to N. Diversion Channel outfall
B3	398–422	2.4 (3.9)	N. Diversion Channel outfall to ABCWUA diversion dam
B4	422–463	4.0 (6.4)	ABCWUA diversion dam to Montaño Bridge
Mo1	463–494	3.0 (4.8)	Montaño Bridge to Interstate 40 Bridge
Mo2	494–528	3.5 (5.6)	Interstate 40 Bridge to Bridge Boulevard Bridge
Mo3	528–575	4.5 (7.2)	Bridge Boulevard Bridge to Tijeras Arroyo (S. Diversion Channel)
Mo4	575–623	4.5 (7.2)	Tijeras Arroyo (S. Diversion Channel) to Interstate 25 Bridge
Mo5	623–657	3.5 (5.6)	Interstate 25 Bridge to Isleta Diversion Dam
Angostura Reach	298–339	35.0 (56.1)	US HWY 550 to Isleta Diversion Dam

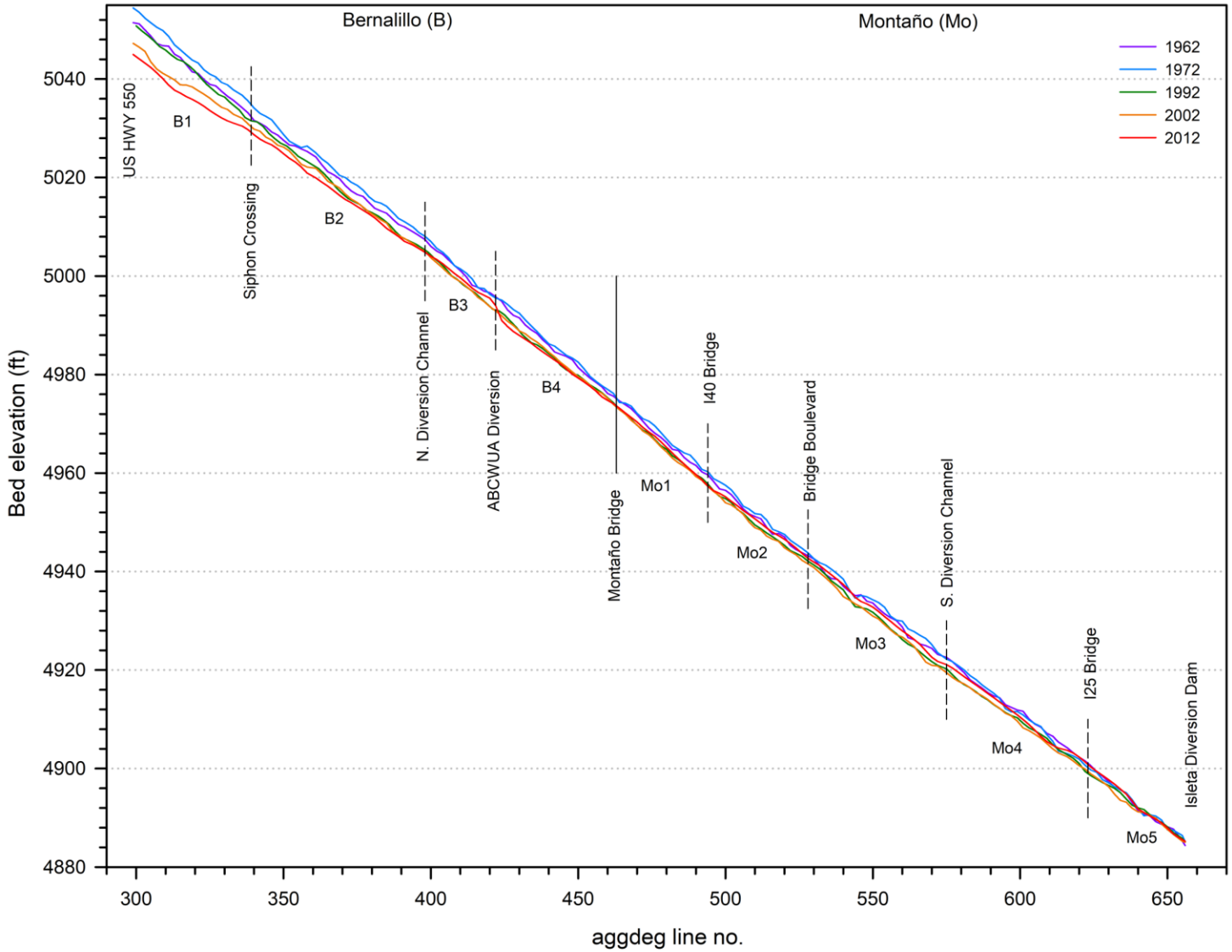


Figure 4. Longitudinal profile of the Angostura Reach including subreach boundaries. Lines represent survey periods: 1962, 1972, 1992, 2002, 2012. Bed elevations (NAVD88) were averaged at the midpoint of every three cross-sections (aggdeg lines) to smooth lines.

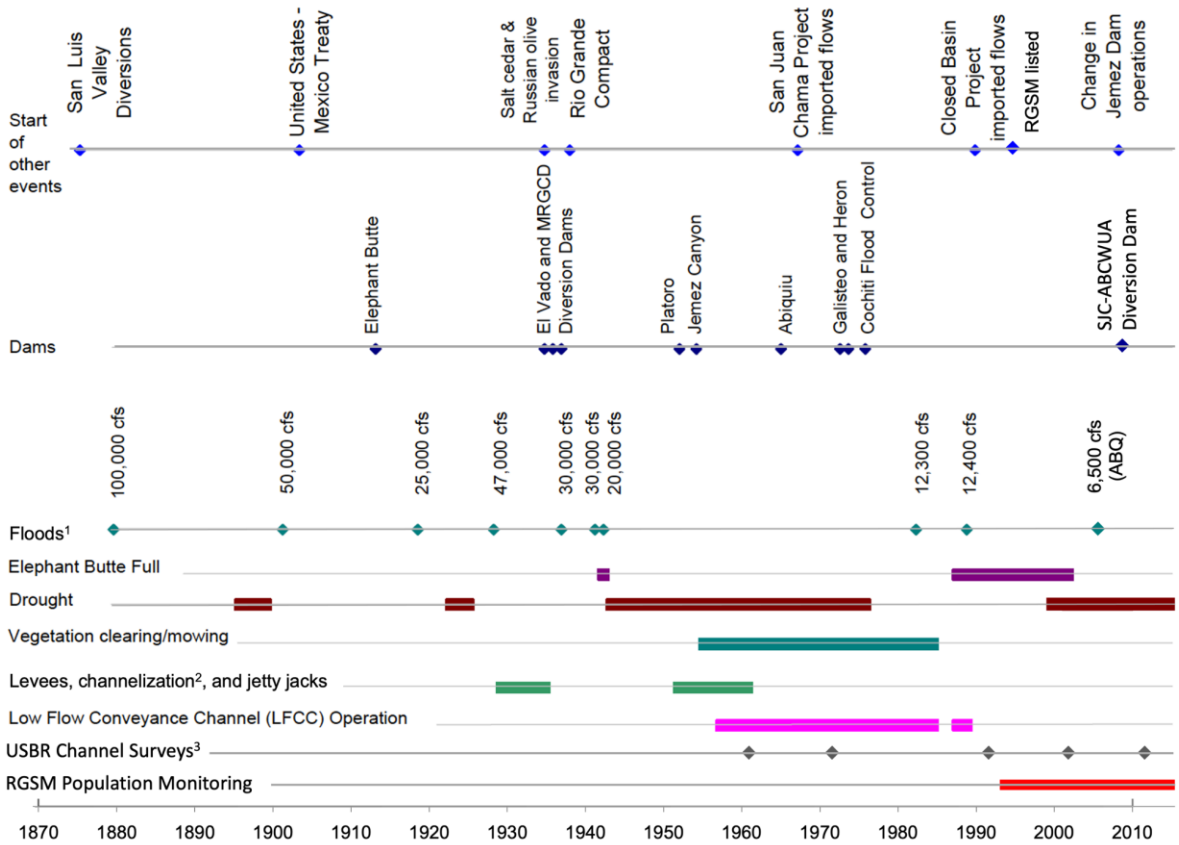
Historical Impacts

The present Middle Rio Grande is largely the result of extensive river engineering and development during the 20th century. While irrigation systems and practices can be traced back to Native American pueblos (ca. 1300–) and Spanish colonization (ca. 1600–), it was not until the late 1800s that American settlers rapidly developed the region's economy by expanding agriculture and livestock production (Scurlock, 1998). However, the Rio Grande was a source of constant hardship for its human inhabitants. Droughts occurred nearly every decade since the 1600s, and major floods (>10,000 cfs) occurred every few years from 1849 to 1942, frequently killing people, livestock, and crops while destroying houses, farmland, and irrigation networks (Scurlock, 1998). Naturally high sediment loads were further exacerbated by upland activities (e.g., forest clearing, overgrazing, development), which rapidly aggraded the river channel, raised the water table, and waterlogged fields, further impeding efficient agricultural production. In response to these challenges, governmental agencies (from local to federal) began implementing engineering solutions.

The extensive construction of levees, irrigation networks, diversion dams, water storage facilities, and conversion of floodplain to farmland was initiated by the Middle Rio Grande Conservancy District (MRGCD) starting in the 1920s. Initial efforts targeted flood control via channel modifications (i.e., jetty-jack lines) and construction of spoil levees, which drastically reduced floodplain size and connectivity. Additional modifications for agriculture (e.g., water diversion structures, irrigation channels, and riverside drains) altered surface and ground water hydrology. The ambitious scope of MRGCD projects and ongoing challenges required them to seek direct logistical and financial assistance of the federal government (the U.S. Bureau of Reclamation [USBR] and the U.S. Army Corps of Engineers [USACE]). In 1950, the Middle Rio Grande Project was approved by Congress and involved major engineering initiatives by USBR and USACE including: (1) flood and sediment control on the mainstem Rio Grande (Cochiti Dam; 1973), (2) flood and sediment control dams on tributaries (Jemez Canyon [1953]; Galisteo [1970]), (3) rehabilitation of mainstem diversion dams (Angostura, Isleta, San Acacia), and (4) construction of the Low Flow Conveyance Channel (LFCC [1951]). While the Middle Rio Grande Project mostly relieved the hardships of drought and flooding, it has contributed to the ecological stress on the Rio Grande ecosystem, largely disconnected the river from its floodplain, and has imperiled multiple species including the Rio Grande Silvery Minnow. The hydrologic and geomorphic changes to the Middle Rio Grande have been substantial (Figure 5, Table 2). Habitat degradation resulting from Middle Rio Grande Project activities is driven largely by modifications to the natural flow and sediment regimes, channelization of the river, and fragmentation of the Middle Rio Grande.

Native Ichthyofauna of the Middle Rio Grande

The Rio Grande ecosystem was historically characterized by highly variable and often harsh environmental conditions (e.g., spring flooding, high suspended sediment concentrations, high-intensity precipitation events, drought periods). Persistence of native fishes was facilitated by specialized life history strategies adapted to these conditions. For example, the Rio Grande Silvery Minnow *Hybognathus amarus* produces non-adhesive, nearly neutrally-buoyant eggs that are passively dispersed by water currents – a reproductive mode referred to as pelagic-broadcast spawning (Platania and Altenbach, 1998; Worthington et al., 2018). This reproductive strategy is adapted to the regional hydrologic conditions that characterize the Great Plains of North America – seasonally predictable, highly-variable periods of high discharge and sediment loading during the spring. Historically, the Middle Rio Grande supported five native pelagic-broadcast spawning fishes: the Phantom Shiner *Notropis orca* and the Rio Grande Bluntnose Shiner *Notropis simus simus* are extinct, the Speckled Chub *Macrhybopsis aestivalis* and the Rio Grande Shiner *Notropis jemezanus* are extirpated, and the Rio Grande Silvery Minnow is the only extant, but imperiled species (Bestgen and Platania, 1990, 1991). Historically, the Rio Grande Silvery Minnow was abundant in the Rio Grande from Española, NM to the Gulf of Mexico, including the Pecos River from Santa Rosa, NM to its confluence with the Rio Grande (ca. 2,400 mi [3,900 km]). Currently, this species occurs solely in the Middle Rio Grande (ca. 250 km [155 mi]), which is less than ten percent of its historical range. The loss of multiple native pelagophilic species and the decline of the Rio Grande Silvery Minnow serves as an indicator of the ecological consequences of hydrologic and geomorphic alteration of the Rio Grande over the past century.



[1] Flood events listed were measured at various locations between Otowi and San Marcial.
[2] Channelization includes temporary channel construction 2000–2004.
[3] Channel surveys used in this study.

Figure 5. Timeline of substantial hydrologic and geomorphic impacts to the Middle Rio Grande, 1870–2015. Figure modified from Makar and Aubuchon (2012).

Rio Grande Silvery Minnow Biology and Habitat Syntheses

The Rio Grande Silvery Minnow is a relatively small and short-lived minnow of the cyprinid family (Figure 6). Wild fish are generally 30–60 mm standard length (SL), depending on their age and the time of year, and adults may reach up to 90 mm SL (USFWS, 2010; Horwitz et al., 2018). The typical lifespan is one to two years in the wild – the abundance of older age classes generally declines through summer and autumn, suggesting high incidence of mortality after spawning (Horwitz et al., 2018; Dudley et al., 2022). Consequently, newly spawned individuals dominate the population (>95%) given adequate spring spawning flows that year (Horwitz et al., 2018). This species exhibits rapid growth, attaining morphological development by autumn (i.e., juvenile life-stage) and reproductive maturity in <12 months (i.e., adulthood; Figures 7–8). Thus, given favorable environmental conditions (e.g., elevated and prolonged spring runoff, low intermittency during summer), substantial population increases can be observed within a year (Dudley et al., 2022). Conversely, its short lifespan heightens risk to substantial population declines, which can occur during just one year with poor hydrologic conditions (Dudley et al., 2022). The interaction of flow, channel morphology, and habitat conditions are strongly related to the population dynamics of this species (Figures 7–10).

Spawning occurs between mid-April and mid-June, with peak spawning typically early to mid-May, which historically coincided with seasonally predictable, yet highly variable snowmelt runoff from mountainous headwaters. Spawning appears to be stimulated by increases in flow and water temperature during spring (Figure 8). This species has a distinct egg type that is unique within the Middle Rio Grande fish community. Females release relatively large (~3.5 mm), nearly neutrally buoyant (specific gravity ~1.005), non-adhesive eggs that are suspended in the water column by trace currents (<1 cm/s) and high-suspended sediment concentrations but settle to the bottom without some sustained vertical turbulence (Platania and Altenbach, 1996; Dudley and Platania, 2007; Medley and Shirey, 2013). Whereas most freshwater fishes produce eggs that minimize displacement (i.e., adhesive, dense eggs), eggs of this species are susceptible to downstream displacement (i.e., drift), a key aspect of their early life history (Dudley and Platania, 2007). This reproductive strategy facilitates rapid dispersal of propagules to favorable habitats and allows spawning to occur early in the year when conditions are often harsh (e.g., peak flows, high sediment loads), thus maximizing the time available for growth and development before the onset of winter, when colder water temperatures decrease rates of growth and activity. Fecundity (i.e., number of spawned eggs) of this species is high (2,000–10,000+ eggs; Caldwell et al., 2018); high fecundity is associated with species that experience high rates of mortality during early life-stages (i.e., Type III survivorship). Egg hatching occurs within 24–48 hours and is influenced by water temperature (Platania, 2000). Eggs and newly hatched larvae drift until retained in low or trace water velocity habitats (i.e., floodplains, backwaters, and shorelines); eggs and larvae can also be displaced from the river entirely (e.g., drift into reservoirs). The lateral and longitudinal displacement of eggs and larvae increases the likelihood that propagules will reach nursery habitats (i.e., habitats favorable for growth of larvae).

The larval life-stage is arguably the most critical and sensitive phase of fish development. Larvae lack the physical size and morphological definition (i.e., fins and rays), sensory capabilities, and learned behaviors of adults, all of which influence their survival. Larvae are particularly limited in swimming ability, and therefore, depend on the availability of shallow, low velocity habitats. Larval Rio Grande Silvery Minnows require about 4–10 days to develop free-swimming ability (i.e., ability to move horizontally) and considerably longer (~50 d) to reach the juvenile life-stage (Platania, 2000). Floodplain inundation increases the availability of nursery habitats and given a sufficient duration of spring flooding, increases the likelihood newly spawned fish will survive harsher conditions in the main channel (e.g., higher water velocities, competition, predators) when spring runoff recedes.

The Rio Grande Silvery Minnow is typically encountered in shallow, low velocity habitats (e.g., <60 cm, <40 cm/s). Habitat use is likely determined by physiological constraints associated with small-bodied fishes. Although this species is most frequently collected in low-velocity habitats, adults are capable of swimming at higher velocities and long distances — swimming performance studies have demonstrated swimming speeds of 100–118 cm/s for short intervals (5–15 s) and the capability to swim 5–125 km in <72 h (Bestgen et al., 2010). Field observations have verified extensive upstream movements in the wild (>20 km; Archdeacon and Remshardt, 2012; Platania et al., 2020). The early life history of this species (e.g., drifting eggs and larvae) suggests that movement and redistribution of individuals (i.e., dispersal) is necessary for long-term population persistence.

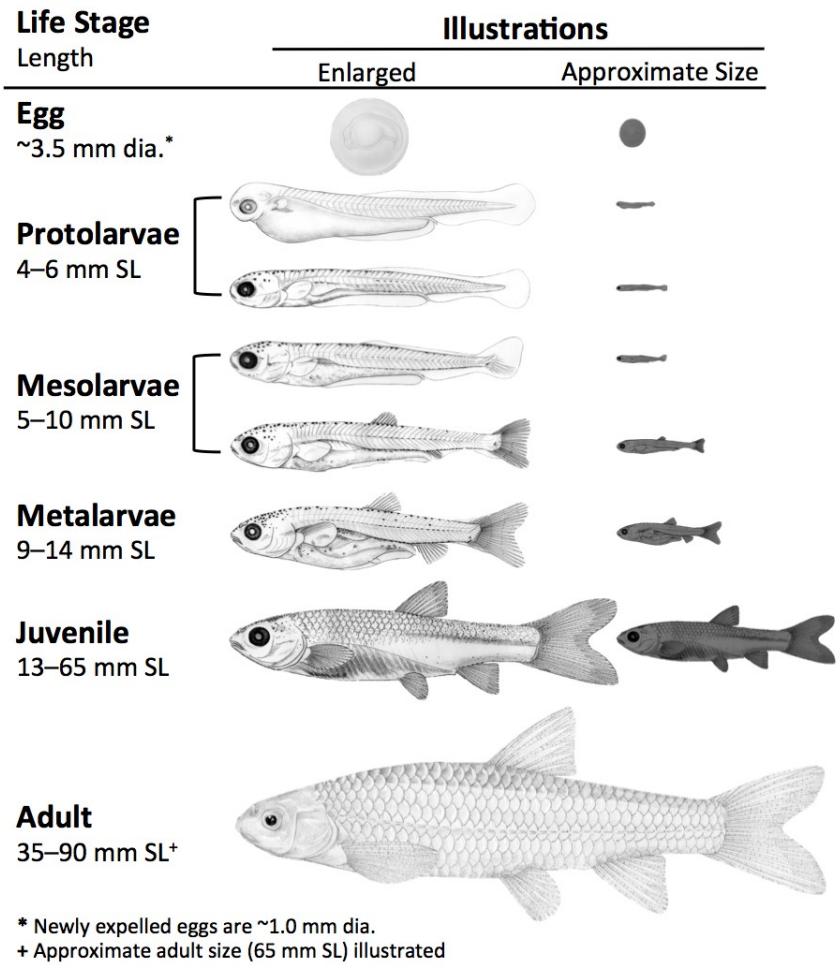


Figure 6. Life-stages of the Rio Grande Silvery Minnow and their approximate actual sizes. Lengths were obtained from Brandenburg et al. 2018 (egg, larvae, juvenile); USFWS 2010 and Horwitz et al. 2018 (juvenile, adult). Note considerable overlap of lengths especially between juvenile and adult. Illustrations by J.P. Sherrod (egg) and W.H. Brandenburg (larvae, juvenile, adult).

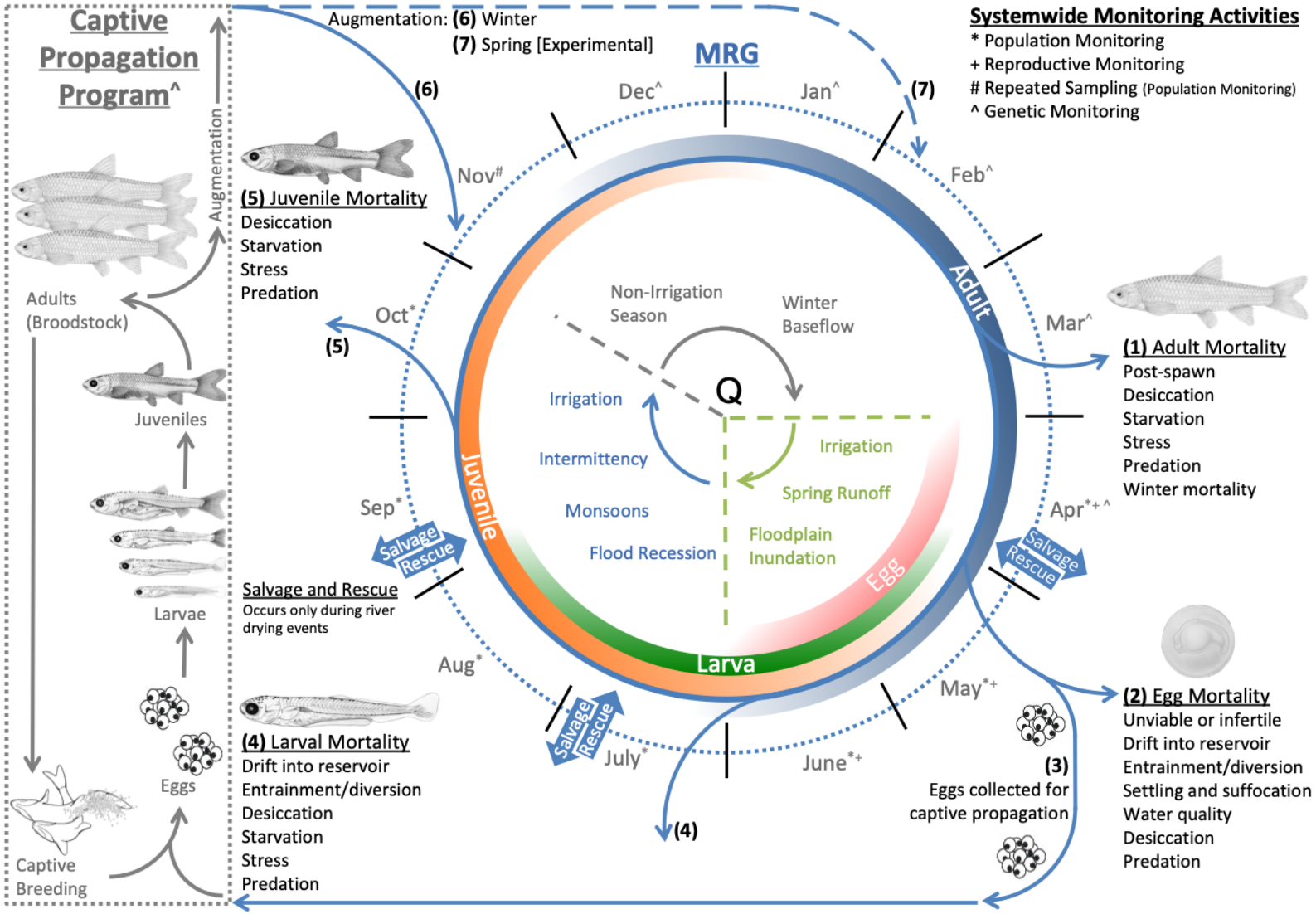


Figure 7. Conceptual life history model of the Rio Grande Silvery Minnow. Life history is largely driven by streamflow (Q at center) over the course of one year. The approximate timing and duration of life-stages are shown as concentric colored bars. Arrows indicate removal and addition to the population through potential sources of mortality and interaction with recovery efforts (i.e., captive propagation, salvage).

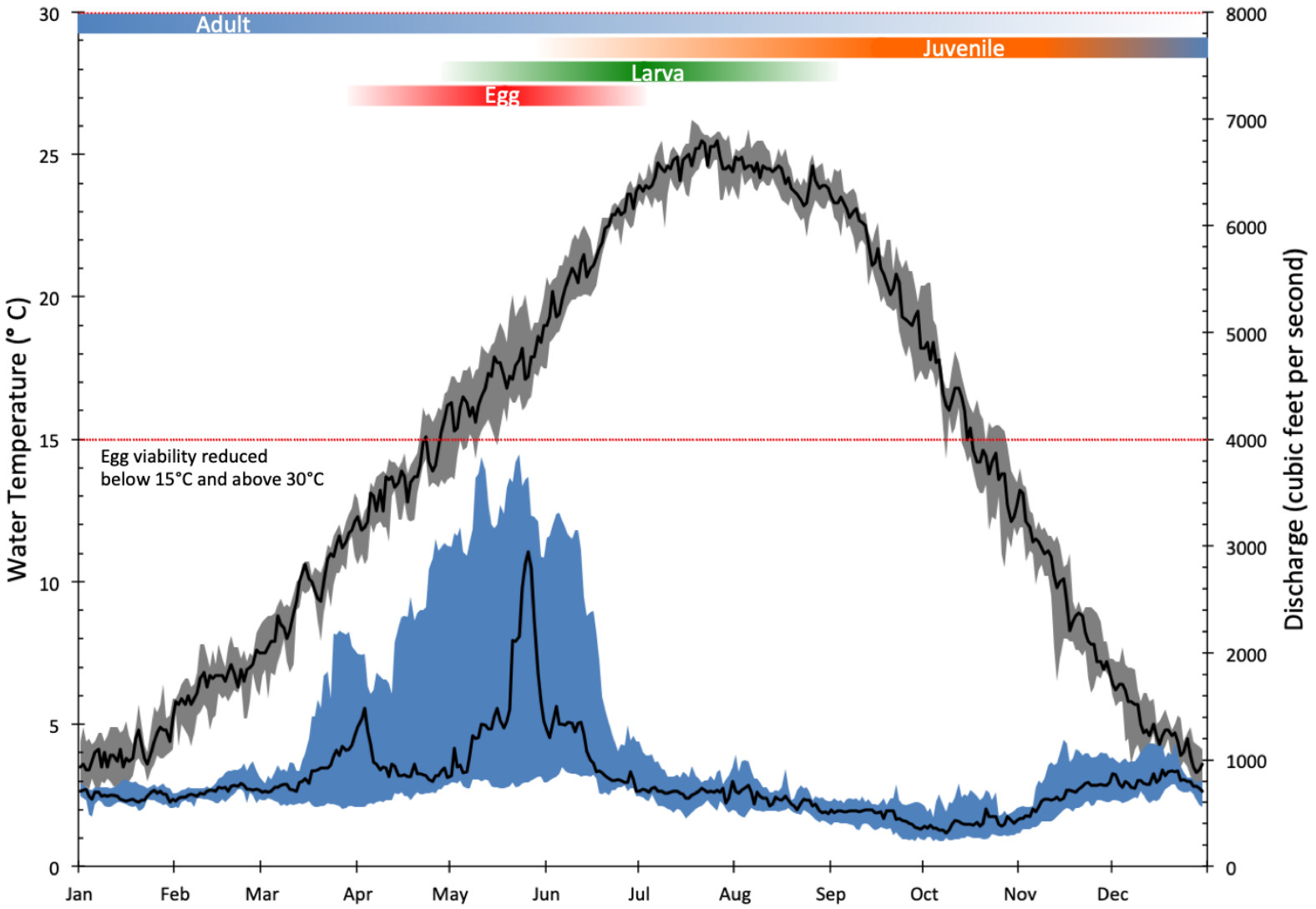


Figure 8. Hydrograph and thermograph for the Rio Grande at Alameda, NM (USGS 08329918). Black lines represent median daily discharge and water temperature (Q_{50}; T_{50}) and shaded regions (Q [blue]; T [gray]) represent interquartile ranges (Q_{25} to Q_{75}; T_{25} to T_{75}) for the period 2006–2018. Colored bars indicate the approximate timing and duration of the primary life-stages of the Rio Grande Silvery Minnow. Red lines indicate approximate thermal thresholds for egg viability.

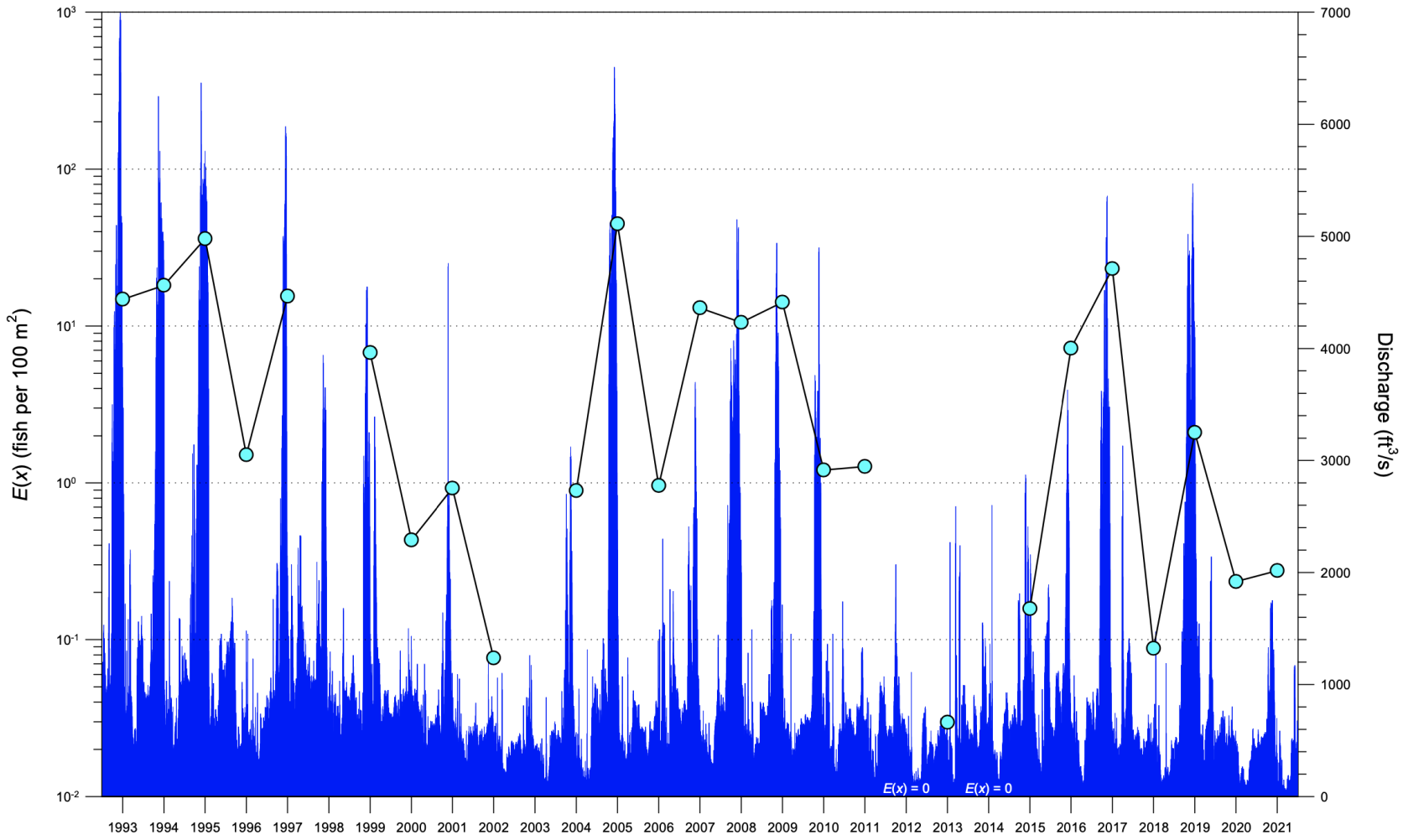


Figure 9. Densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data from all sites [MRG]) and mean daily discharge data from the Albuquerque Gage across years (from Dudley et al. 2022). Sampling did not occur in 1998, and $E(x)$ could not be computed for 2003.

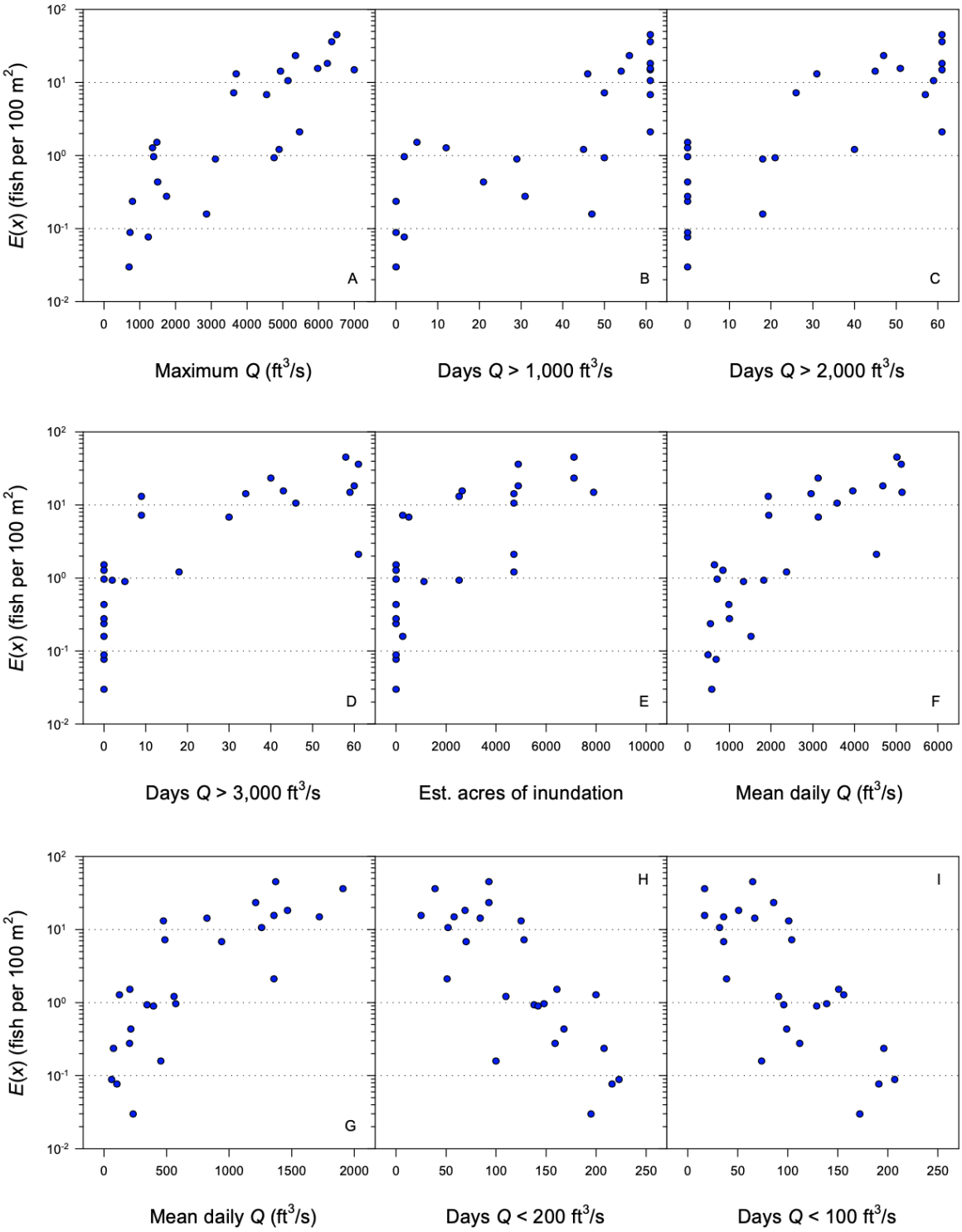


Figure 10. Bivariate plots of densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data from all sites [MRG]) and seasonal hydrologic metrics 1993–2021 (Albuquerque Gage USGS 08330000 data [A–F, spring runoff metrics], and San Marcial Gage USGS 08354900 data [G–I, summer low flow metrics]; from Dudley et al. 2022).

Implications for Habitat and Population Dynamics of the Rio Grande Silvery Minnow

Hydrologic and geomorphic alteration of the Middle Rio Grande has negatively impacted habitat and population dynamics of the Rio Grande Silvery Minnow. The primary factors impacting habitat and population dynamics include flow and sediment regimes, channel modifications and responses, floodplain connectivity, and river fragmentation (Table 2).

Flow and sediment are the main drivers of geomorphic change on the Middle Rio Grande (Figure 11). Flow characteristics have changed relative to historical conditions, in particular, the magnitude, frequency, and duration of peak spring flows have decreased (Swanson et al., 2011; Blythe and Schmidt, 2018). Peak flows and flooding during spring are important for spawning and survival of early life-stages of the Rio Grande Silvery Minnow (Mortensen et al., 2019). Additionally, the duration and frequency of low flows have increased, and considerable distances can dry during summer. Prolonged low flows elevate mortality of juvenile and adult Rio Grande Silvery Minnows (Archdeacon, 2016). The abundance of the Rio Grande Silvery Minnow has been closely linked to spring runoff events and drought periods (Figures 9–10). Concurrent with hydrologic impacts, natural sediment inputs have been extensively disconnected (e.g., Cochiti Dam; Figure 12) to reduce historical trends of channel aggradation. Current sources of sediment are primarily from the erosion of the streambed and banks and ephemeral tributary inputs (Makar, 2010; Posner, 2017). The sediment regime is intimately linked with the flow regime, and together, they historically sustained ecosystem integrity by maintaining physical habitat conditions and providing ecologically significant disturbance events (Wohl et al., 2015).

Channel modifications and altered flow-sediment regimes have transformed the river from a wide, shallow, braided planform to a relatively narrow, single-threaded, incised channel (Makar, 2010; Swanson et al., 2011). Large-scale channelization in the 1950s involved the installation of jetty jacks to stabilize banks and protect levees (Figure 13; Berry and Lewis, 1997; Grassel, 2002). Jetty jack lines initially caused rapid narrowing (~25 ft/year) of the river channel (Grassel, 2002; Swanson et al., 2011). Today, narrowing continues at a lesser rate, largely driven by the encroachment of riparian vegetation during prolonged low flow periods (Makar, 2010). Changes to channel width and depth have been dramatic (Figures 14–16), causing negative impacts on aquatic habitats. Channel narrowing has reduced the availability and complexity of habitat features within the river. Also, increased water velocities and depths resulting from narrowing and incision further reduce the amount of suitable habitat available for the Rio Grande Silvery Minnow (e.g., LaForge et al. 2020; Yang et al., 2020). Channel incision has also reduced floodplain connectivity (Massong et al., 2006).

Floodplain connectivity has been reduced by changes to the river channel (i.e., channelization and incision) and flow regime (i.e., reduced spring runoff characteristics). Floodplains are critical to early life stages of the Rio Grande Silvery Minnow as these areas provide habitats that facilitate the retention of eggs and larvae. Current peak flows during spring runoff do not typically overbank substantially as they did historically, but rather produce advective conditions within the main channel that are capable of rapidly displacing eggs and larvae downstream to unsuitable habitats (i.e., Elephant Butte Reservoir; Dudley and Platania, 2007; Widmer et al., 2012). Spawning without sufficient floodplain connectivity results in higher egg passage rates, indicating low egg retention rates upstream (Dudley et al., 2018). Spring flooding also increases food resource availability through nutrient enrichment and increased productivity (i.e., algae and small invertebrates; Valett et al., 2005; Kennedy and Turner, 2011) that facilitate rapid growth and development. Without the egg retention mechanisms or optimal nursery habitats provided by floodplains, early life-stages are likely to experience high mortality rates.

Fragmentation by dams inhibits egg retention mechanisms and restricts population movement and redistribution within the river. The likelihood eggs and larvae will drift into unsuitable habitats is increased in fragmented systems (Dudley and Platania, 2007; Perkin and Gido, 2011; Hoagstrom, 2015; Perkin et al., 2015). Few fish transported past diversion dams are able to return upstream, contributing to net downstream displacement of offspring and typically higher abundances in downstream reaches (e.g., Dudley et al., 2018; 2019). The Rio Grande Silvery Minnow and other pelagic-broadcast spawning fishes require upstream dispersal for long-term persistence in upstream reaches (Speirs and Gurney, 2001; Humphries and Ruxton, 2002; Platania et al., 2020). Additionally, connectivity between upstream and downstream populations maintains genetic diversity and viability in the wild population (Osborne et al., 2012; Carson et al., 2020).

Table 2. Summary of hydrologic and geomorphic impacts to the Middle Rio Grande and their implications for the population of the Rio Grande Silvery Minnow.

Factor	Causes	Hydrologic and Geomorphic Impacts	Implications for Population Dynamics[4]
Flow Regime	• Mainstem and tributary dams • Water use, storage, and flood control • Agricultural development (riverside drains) • Precipitation variability (wet and dry periods)	• Reduced magnitude, duration, and frequency of spring flood events • Increased duration and frequency of low flows (intermittency) • Colonization by riparian vegetation (low flows)[3] • Surface water – groundwater connectivity (increased depth to GW)	• Reduced availability and persistence of spawning and nursery habitats • Reduced availability of slackwater habitats for retention of eggs and larvae • Reduced availability, persistence, and quality of refuge habitats during low flows • Increased mortality during river intermittency
Sediment Regime	• Mainstem and tributary dams • Reduced sediment supply	• Channel incision (degradation) • Bed coarsening/armoring	• Reduced habitat complexity and availability • Food resource availability (substrates available for algal growth)
Channel modifications, narrowing, aggradation/ degradation[1]	• Channelization[2] • Mainstem and tributary dams • Reduced spring runoff • Reduced sediment supply • Colonization by riparian vegetation(low flows)[3]	• Single-threaded, laterally confined channel • Loss of instream and riparian areas • Channel incision (degradation) • Channel narrowing • Increased bankfull discharge • Perched channel (aggradation) • Depth to groundwater (aggradation)	• Reduced habitat complexity and availability • Reduced availability and persistence of spawning and nursery habitats • Reduced availability of slackwater habitats for retention of eggs and larvae • Reduced availability, persistence, and quality of refuge habitats during low flows • Increased mortality during river intermittency
Floodplain Connectivity	• Mainstem and tributary dams • Channelization[2] • Channel incision (degradation)	• Reduced magnitude, duration, and frequency of spring flood events • Laterally confined, incised channel • Increased bankfull discharge	• Reduced availability and persistence of spawning and nursery habitats • Reduced availability of slackwater habitats for retention of eggs and larvae
Fragmentation	• Mainstem and tributary dams	• Reduced upstream-downstream hydrologic connectivity	• Limited population movement and redistribution between reaches • Net displacement of eggs and larvae downstream • Reduced genetic diversity and long-term population persistence

[1] Channel degradation occurs in the all reaches of the Middle Rio Grande, however, aggradation occurs in the San Acacia Reach.
[2] Channelization refers to the installation of jetty jacks and levees along the Middle Rio Grande.
[3] Riparian vegetation contributes to channel narrowing during prolonged low flow periods (droughts), however, increased roughness on vegetated surfaces might result in marginal increases in the availability of slackwater habitats.

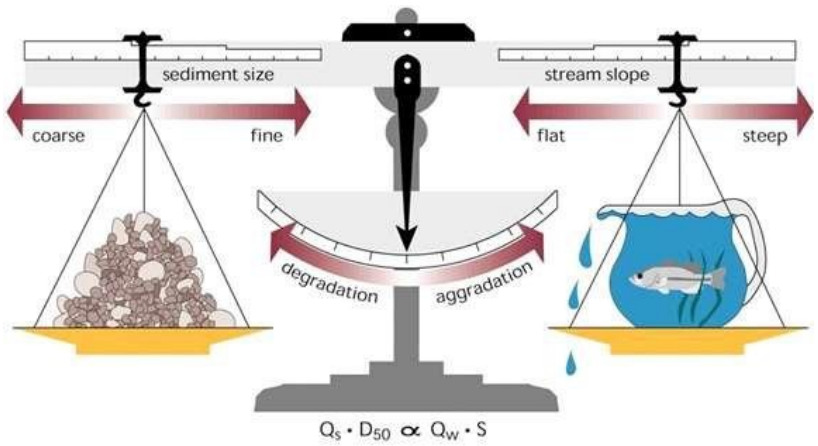


Figure 11. Illustration of Lane's Balance showing the relationship between sediment discharge (Q_s), median sediment size (D_{50}), water discharge (Q_w), channel slope (S), and aggradational or degradational responses to disturbance. Relationship developed by Lane (1955) and enhanced graphics by Rosgen (1996).

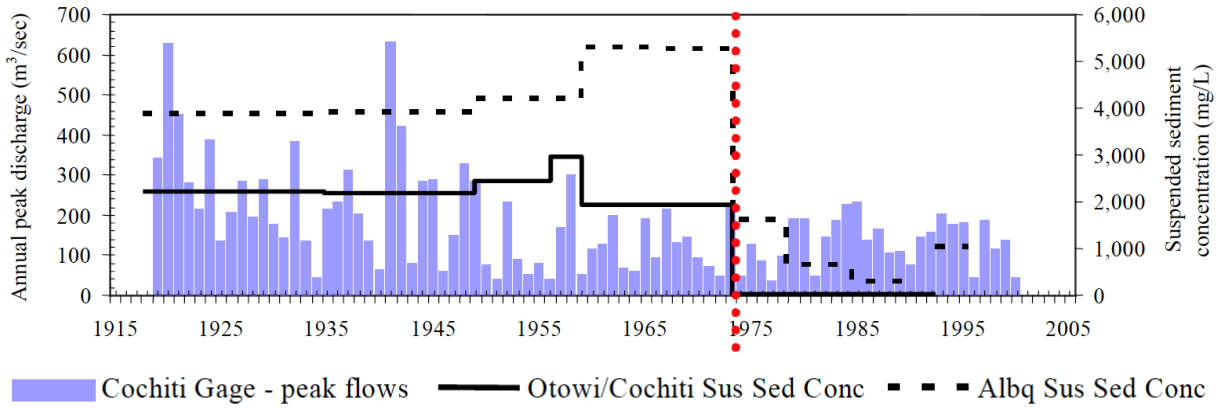


Figure 12. Summary of changes to flow and sediment regimes in the Middle Rio Grande 1918–2000 (from Richard and Julien, 2003). Construction of Cochiti Dam is indicated by the red dotted line (1973). Note higher peak flows and suspended sediment concentrations prior to closure of Cochiti Dam.

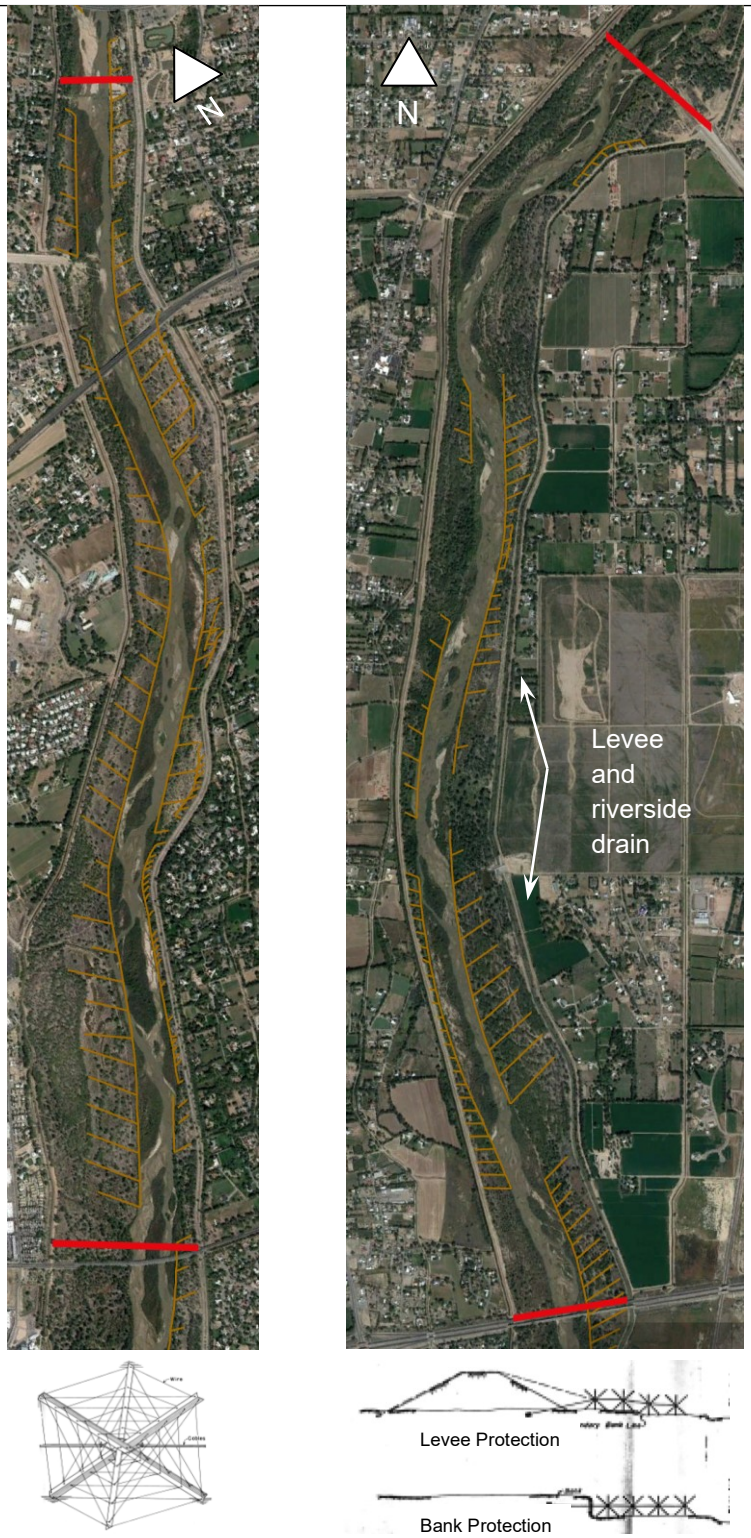


Figure 13. Example of channel modifications in the Angostura Reach. Orange lines show jetty jack lines (1962) and red lines show subreach boundaries. Locations of levees and riverside drains are visible. Left panel shows subreach B4 (4.0 river mi); right panel shows subreach Mo4 (4.5 river mi). Example jetty jack (left) and line placement (right) are illustrated at bottom (Grassel, 2002). Aerial imagery obtained from GoogleEarth.

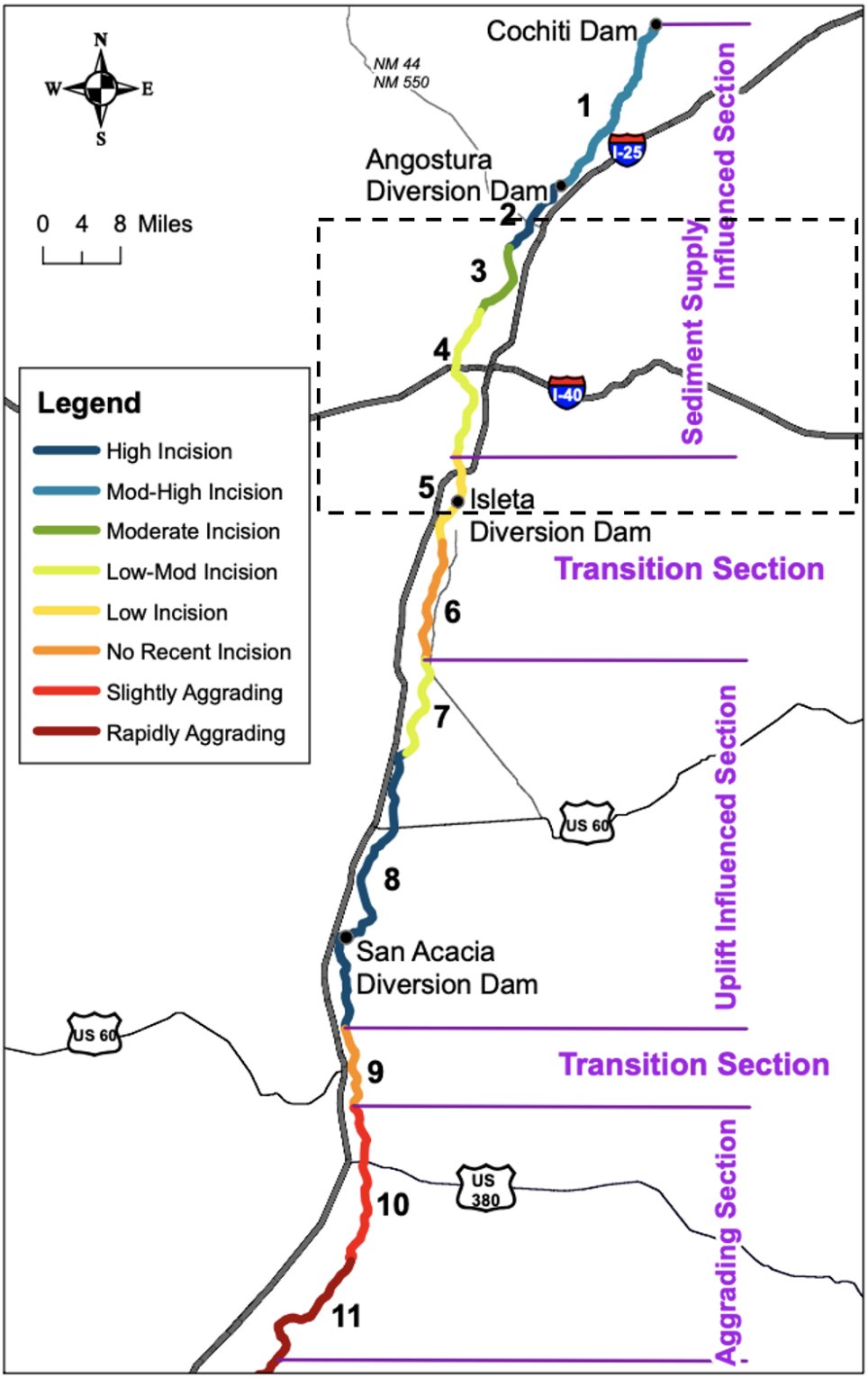


Figure 14. Middle Rio Grande aggradation-degradation trends 1936–2002 (from Massong et al. 2006). Trends are described spatially as: high incision (>6 ft), moderate incision (3–5 ft), low incision (<3 ft), slightly aggrading (<10 ft), and rapidly aggrading (>10 ft). Dashed black line shows the Angostura Reach (sediment supply influenced section).

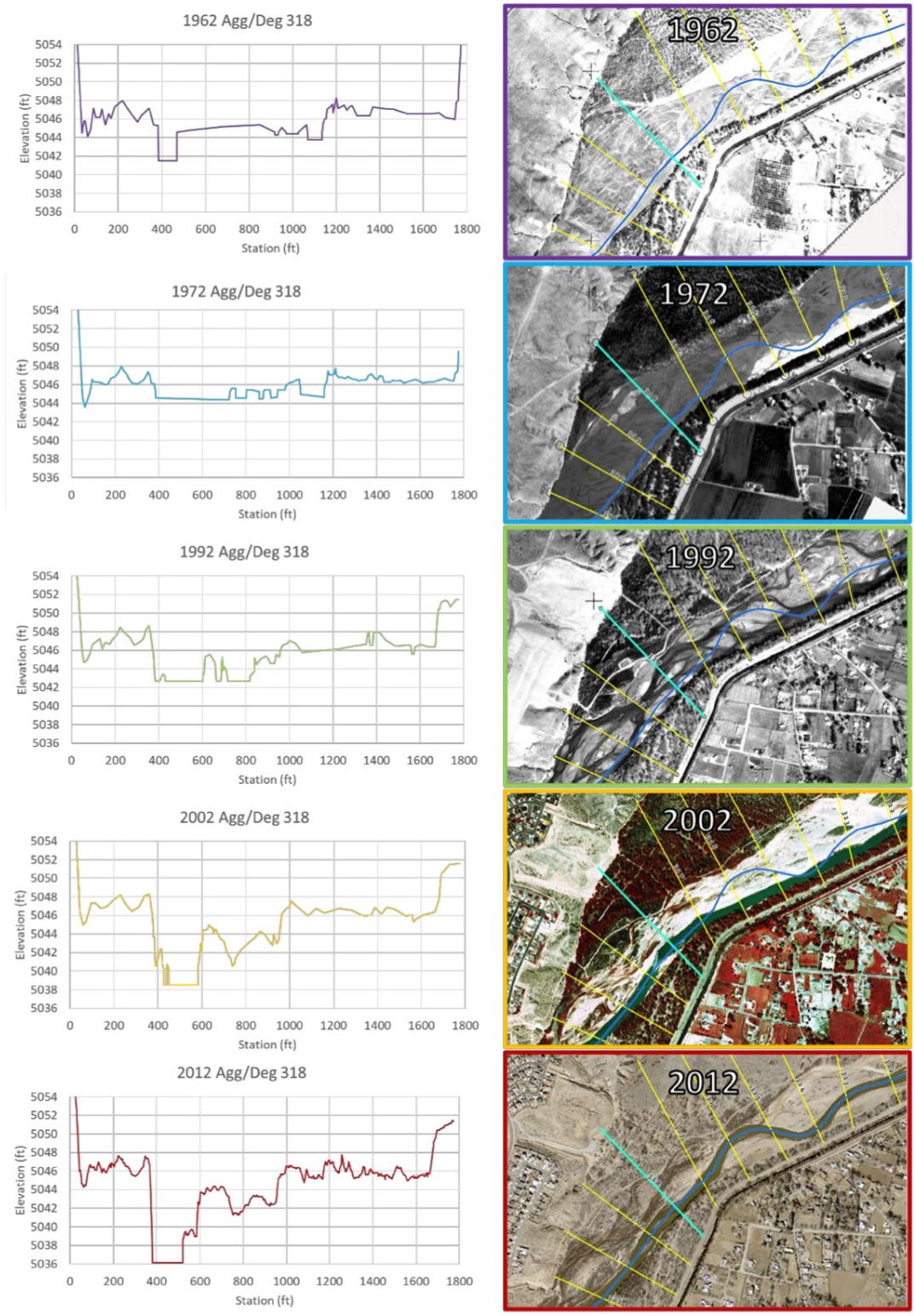


Figure 15. Example of channel narrowing and incision in the Angostura Reach (from Radobenko et al., 2023). Panels (top to bottom) illustrate changes to the cross-section (left) and planform (right) at agg/deg line 318 (subreach B1; just downstream of US HWY550 bridge) between 1962 and 2012.

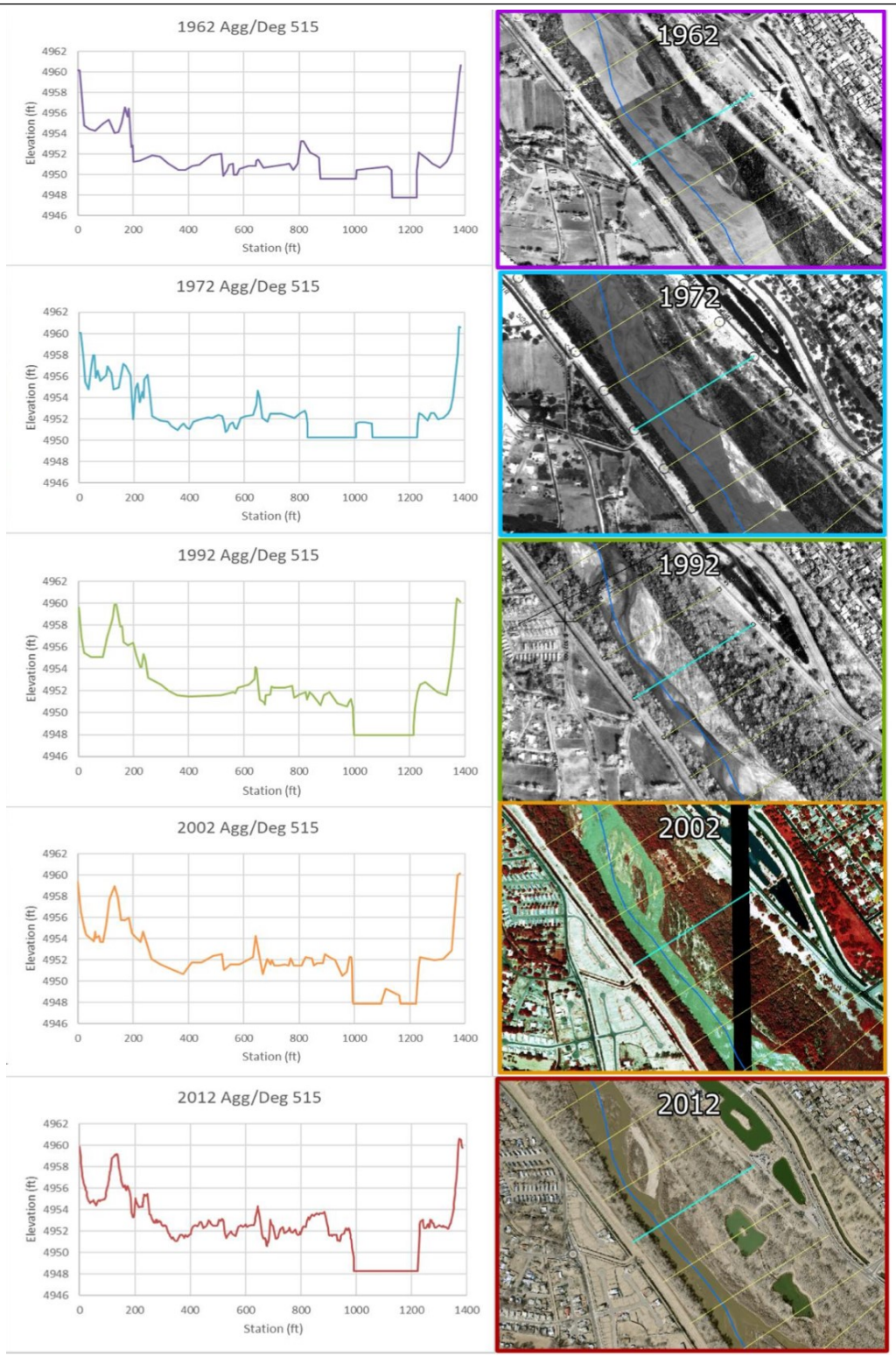


Figure 16. Example of channel narrowing and incision in the Angostura Reach (from Anderson et al., 2023). Panels (top to bottom) illustrate changes to the cross-section (left) and planform (right) at agg/deg line 515 (subreach Mo2; just downstream of Central Ave. bridge) between 1962 and 2012.

DATA AND METHODS

This section describes the primary datasets and analytical techniques used to assess process-linkages among morpho-dynamics, habitat availability, and the Rio Grande Silvery Minnow population of the Middle Rio Grande.

Data

Channel Geometry (U.S. Bureau of Reclamation)

The U.S. Bureau of Reclamation has systematically surveyed channel cross-sections along the Middle Rio Grande. Since 1962, channel cross-sections have been estimated using photogrammetry at fixed transects known as aggradation/degradation (agg/deg) lines – cross-sections of the channel and floodplain that are spaced at approximately 500 ft intervals along the length of the Middle Rio Grande (Varyu, 2013). Surveys have occurred approximately every 10 years (1962, 1972, 1992, 2002, 2012; Figure 5). All elevation data are in NAVD88. The period 1962–2012 includes aerial imagery and LiDAR data is also available for recent years (2012). These surveys are used to estimate sedimentation and morphological trends in the river channel and floodplain.

Channel-floodplain elevations were obtained by aerial surveys and photogrammetric techniques were used to estimate elevations along agg/deg lines for 1962–2002 datasets. 2012 elevation data were obtained by LiDAR and photogrammetry (Varyu, 2013). The agg/deg lines provide relatively accurate cross-sections for areas above and beyond the water surface – below the water surface, an idealized trapezoidal cross-section was estimated using a one-dimensional hydraulic model (HEC-RAS) and the measured discharge during the survey (Holmquist-Johnson and Makar, 2006; Varyu, 2013). The resulting cross-sections form the channel geometries used in HEC-RAS.

Stream Gaging Stations (U.S. Geological Survey)

The U.S. Geological Survey maintains several stream gaging stations in the study area (Figures 2–3, Table 3). The USGS National Water Information System was used to access available stream gage data, which primarily included measurements of discharge, suspended sediment, and water temperature. Although several stream gages are present in the study area, these gaging stations vary in terms of parameters and periods of record (i.e., not all parameters are measured consistently spatially or temporally). Flow statistics and representative hydrographs are shown in Figure 17. Supplementary hydrologic data is included in Appendix A (e.g., raster hydrographs, cumulative discharge curves, precipitation).

Rio Grande Silvery Minnow Population Monitoring Program (ASIR, LLC)

The Rio Grande Silvery Minnow Population Monitoring Program is an ongoing, long-term systematic monitoring study of the Middle Rio Grande fish community conducted since 1993 (Dudley et al., 2022). This effort generates an annual assessment of the abundance and occurrence of the Rio Grande Silvery Minnow (i.e., October sampling), providing a basis for comparing changes in recruitment and survival among years and varying environmental conditions (Figure 5). Fish present in October have survived the cumulative effects of that year's preceding environmental conditions (e.g., spring runoff, monsoons, river drying) and constitute the reproductive cohort heading into the following spring. Further, conditions during October (e.g., streamflow, water temperature, and turbidity) are generally stable and suitable for efficient sampling, as compared to other times of the year (e.g., spring runoff or summer monsoons), making it the most informative month for evaluating long-term population trends.

Specific statistical modeling approaches are required to correctly account for the large proportion of zero values (e.g., zero fish collected during a sampling event) that are typically encountered in studies of rare or imperiled species such as the Rio Grande Silvery Minnow. Mixture models (e.g., combining a binomial distribution with a lognormal distribution) are used to estimate parameters from zero-inflated ecological data such as estimated density, $E(x)$, and probability of occurrence, δ (White, 1978; Welsh et al., 1996; Fletcher et al., 2005; Martin et al., 2005). These parameters were used to assess linkages among morpho-dynamics, habitat availability, and the population of the Rio Grande Silvery Minnow. Additional information on the Rio Grande Silvery Minnow Population Monitoring Program is included in Appendix B.

Table 3. U.S. Geological Survey gaging stations and data availability for the Middle Rio Grande[1]. Gaging stations vary in terms of parameters measured and period of record. Stations listed in black were used for Angostura Reach analyses. Additional parameters (e.g., water temperature, conductivity) are available at several gaging stations. Data were accessed using the USGS National Water Information System.

Station Name	Number	Discharge	Susp. Sediment
Upper Rio Grande			
Rio Grande at Embudo, NM	08279500	1889 – 2021	—
Rio Grande at Otowi Bridge, NM	08313000	1895 – 2021	2008 – 2021
Middle Rio Grande			
Cochiti Reach			
Rio Grande below Cochiti Dam, NM*	08317400	1970 – 2021	1974 – 1988
Galisteo Creek below Galisteo Dam, NM	08317950	1970 – 2021	1971 – 1978
Rio Grande at San Felipe, NM	08319000	1927 – 2021	—
Angostura Reach			
Jemez River below Jemez Canyon Dam, NM	08324000	2009 – 2021	—
North Floodway Channel near Alameda, NM	08329900	1968 – 2021	—
Rio Grande at Albuquerque, NM*	08330000	1942 – 2021	1969 – 2021
Rio Grande at Isleta Lakes near Isleta, NM	08330875	2002 – 2021	—
Isleta Reach			
Rio Grande near Bosque Farms, NM	08331160	2006 – 2021	—
Rio Grande at State HWY346 near Bosque, NM	08331510	2005 – 2021	—
Rio Grande Floodway near Bernardo, NM*	08332010	1957 – 2021	1964 – 2021
Rio Puerco near Bernardo, NM	08353000	1939 – 2021	1955 – 2021
San Acacia Reach			
Rio Grande Floodway at San Acacia, NM*	08354900	1958 – 2021	1959 – 2020
Rio Grande at Bridge near Escondida, NM	08355050	2005 – 2021	—
Rio Grande above US HWY380 near San Antonio, NM	08355490	2005 – 2021	2011 – 2020
Rio Grande Floodway at San Marcial, NM*	08358400	1949 – 2021	1956 – 2019
Rio Grande Conveyance Channel at San Marcial, NM	08358300	1951 – 2021	1955 – 1994
Rio Grande at Narrows in E. Butte Reservoir, NM	08359500	1951 – 2021	—

[1] Several gages are less reliable than others and therefore might not be used in analyses. Period of record is not continuous for some gages.
* indicates the most frequently used gages (most reliable).
— indicates parameter not measured or recorded.

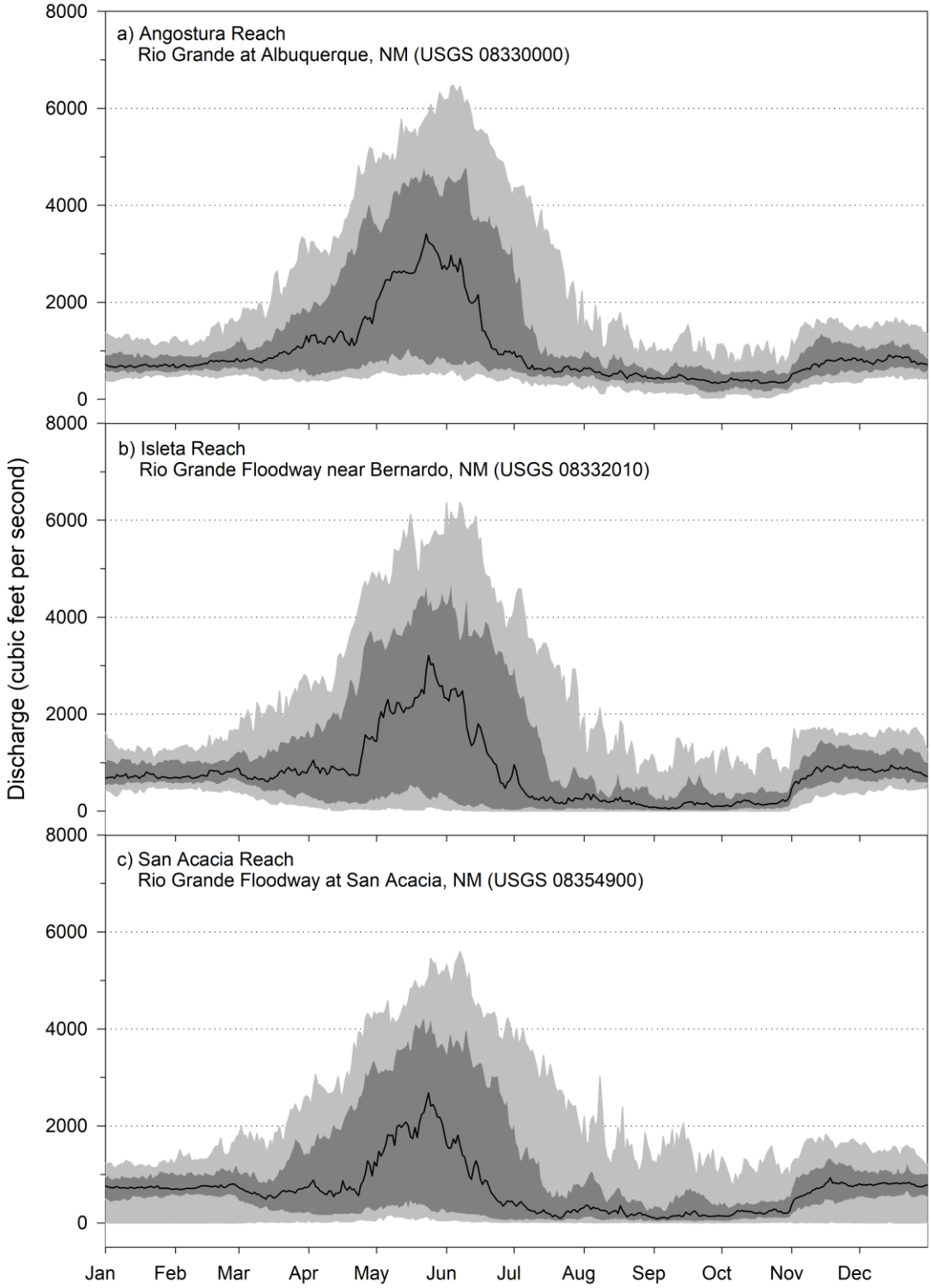


Figure 17. Representative annual hydrographs for Middle Rio Grande reaches: a) Angostura, b) Isleta, and c) San Acacia. Black line shows median discharge (Q_{50}), dark gray shows interquartile range (Q_{25} to Q_{75}), and light gray shows 10th and 90th percentiles (Q_{10}; Q_{90}) 1975–2020.

Methods

Subreach Delineation

Subreaches were delineated in the study area based on physical boundaries, infrastructure locations, or geomorphic characteristics (Figure 3, Table 1). Delineation of these spatial units provided a basis to evaluate how distinct geomorphic characteristics influence key ecological processes (e.g., spring flooding) and habitat conditions, both spatially and temporally (Gurnell et al., 2016). Furthermore, trends identified for subreaches were generalized and related to existing (e.g., Massong et al., 2010; Cluer and Thorne, 2013) and proposed (this report) channel evolution models to improve our understanding of past, present, and future channel conditions, and subsequent impacts to fish habitat.

Subreaches were primarily delineated by inflows (e.g., ephemeral arroyos, agricultural drains), fixed structural features (e.g., bridge crossing, diversion dams), or by cumulative plots of hydraulic variables (e.g., top channel width, flow depth); delineations were made at agg/deg lines where there was a noticeable change in the slope of cumulative plots. Cumulative plots were developed using 2002 and 2012 HEC-RAS model geometry with a discharge of 3,000 cfs; this discharge was selected based on guidance by USBR – it should provide a reasonable approximation of flow conditions within the main channel (i.e., discharge less than overbank). Subreach delineation results for the Angostura Reach are presented in respective reach reports (Radobenko et al., 2023; Anderson et al., 2023).

Hydraulic Modeling (HEC-RAS)

Hydraulic modeling was performed using the Hydrologic Engineering Center – River Analysis System software (HEC-RAS 5.0.6) developed by the U.S. Army Corps of Engineers (USACE, 2016). This software is commonly used in technical river engineering applications to simulate one-dimensional hydraulics in river channels and floodplains.

Bankfull Discharge

In comparison to prior hydraulic modeling analyses of the Isleta and San Acacia Reaches, the estimation of bankfull discharges was not required for the Angostura Reach. The Angostura Reach does not contain perched or semi-perched channels, which pose challenges to accurately modeling inundation patterns using the one-dimensional modeling software, HEC-RAS, and therefore specific modeling procedures were previously implemented to estimate bankfull discharges in the Isleta and San Acacia Reaches (Holste, 2020; Mortensen et al., 2020; 2023). In the Angostura Reach, the range of discharges were modeled without the use of top-of-bank points (i.e., computational levees). Ineffective flow areas, which serve a similar purpose, were implemented in HEC-RAS as needed to improve modeling accuracy (Anderson et al., 2023; Radobenko et al., 2023). Although previously established methods were not used to determine bankfull discharges in this reach, bankfull discharges (1992–2012) were estimated to be greater than 5,000 cfs (Anderson et al., 2023). The selected modeling approaches were deemed reasonable based on the prevailing channel conditions in this reach, however, the potential for differences in results (e.g., flow-habitat curves, TIHMs) to arise across reaches was identified.

Habitat Availability (1962–2012)

A method was developed using HEC-RAS to compare the availability of hydraulically suitable habitats (i.e., water velocity and depth) for the Rio Grande Silvery Minnow across river discharges (500–10,000 cfs) and among channel surveys (i.e., 1992–2012; Doidge et al., 2020). The width-slice method was used to estimate hydraulically suitable habitat areas by processing the velocity and depth distributions in each of the cross-sections (i.e., agg/deg lines). HEC-RAS can analyze lateral flow distribution by cross-section, as described in Chapter 4 of the HEC-RAS Hydraulic Reference Manual (U.S. Army Corps of Engineers, 2016). A cross-section can be divided into a maximum of 45 vertical slices; cross-sections were divided into 45 subdivisions along their width (i.e., width-slices) to characterize the distribution of hydraulic conditions occurring in the channel and floodplain. Hydraulic conditions assessed were hydraulic depth and depth-averaged velocity. Width-slices were distributed to provide a

reasonable approximation of hydraulic conditions across a range of discharge (i.e., main channel [25 subdivisions] and floodplain [20 subdivisions; east and west]). Due to the prevalence of channel incision in this reach (i.e., reduced floodplain connectivity), the number of width-slices in the main channel was increased from prior analyses (i.e., Isleta and San Acacia Reaches) to provide greater resolution. Because floodplains can provide spawning and nursery habitats for the Rio Grande Silvery Minnow and contain more topographic variability than the main channel, 10 width-slices were assigned in each floodplain and 25 width-slices were assigned to the channel, as shown in Figure 18.

A steady flow analysis was run in HEC-RAS for the years 1962, 1972, 1992, 2002, and 2012 for thirteen discharges ranging between 500–10,000 cfs. Flows in the Middle Rio Grande tend to be below 5,000 cfs; therefore, to better represent these flows, increments of 500 cfs were used up to a discharge of 5,000 cfs. After running a steady flow analysis, the flow distribution data were exported to Microsoft Excel for further analysis. For each cross section, the data were analyzed using the width-slice method to estimate areas meeting both the velocity and depth habitat criteria for each life-stage of the Rio Grande Silvery Minnow. Normalized habitat areas (ft^2 per mile) were obtained by summing the width-slices of suitable habitat for each cross-section, multiplying by 500 feet (the approximate spacing of agg/deg lines), and dividing by the length of the reach.

Habitat Mapping (2012)

Habitat maps were generated using the RAS Mapper function of HEC-RAS. This function displays potentially inundated areas using one-dimensional hydraulic modeling results and a digital elevation model. The digitial elevation model was derived from a LiDAR survey, providing ground surface elevations representative of the topography of the riparian area (i.e., main channel and floodplain). RAS Mapper interpolates one-dimensional hydraulic modeling results to a two-dimensional water surface that is distributed across the main channel and floodplain.

LiDAR data obtained in 2012 were used to create the digital elevation model for the Angostura Reach. Using the steady flow data output from HEC-RAS, RAS Mapper distributed depth and velocity values over the digital elevation model for several specified discharges (1,500, 3,000, and 5,000 cfs). A model was developed using the ModelBuilder feature of ArcMap 10.6 (ESRI, 2018) – this model used rasters of depth and velocity from the RAS Mapper output to generate small polygons containing the estimated depth and velocity values for each flow profile (1,500, 3,000, and 5,000 cfs). Polygons were filtered by the hydraulic habitat criteria for each life-stage of the Rio Grande Silvery Minnow to create habitat maps.

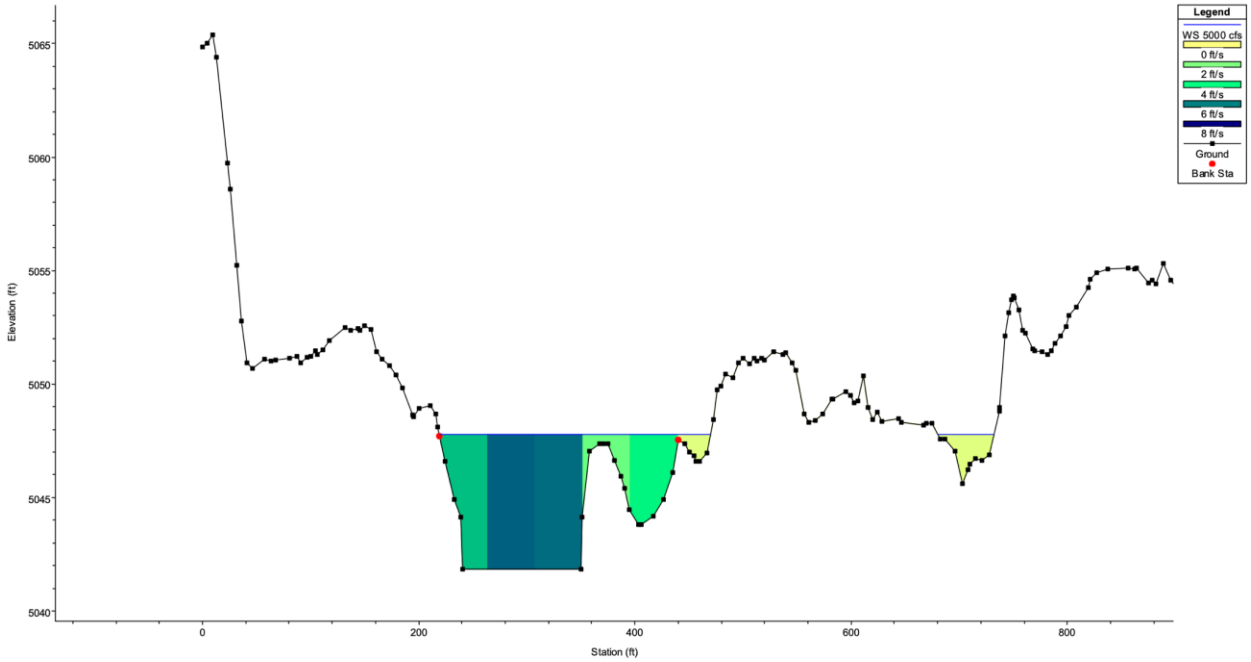


Figure 18. HEC-RAS graphic output showing the lateral velocity distribution at a cross-section (width-slices). Width-slices were distributed with 25 slices in the main channel and 20 slices on the floodplain (n = 45 total slices per cross-section; from Radobenko et al., 2023).

Rio Grande Silvery Minnow Habitat Analyses

Habitat Criteria by Life-Stage

Physical habitat criteria (i.e., velocity, depth) were proposed for the primary life-stages of the Rio Grande Silvery Minnow (Table 4; Figure 19; Mortensen et al., 2019). Physical habitat criteria are commonly used to describe habitat quality, as they are relatively easy to characterize in the field and facilitate application in hydraulic modeling (i.e., HEC-RAS). Habitat criteria were most restrictive for larvae, followed by juveniles and adults, respectively. There was considerable overlap in the hydraulic habitat criteria used by all three life-stages, particularly juveniles and adults. Criteria for adult and juvenile Rio Grande Silvery Minnows were informed by long-term population monitoring efforts, habitat use studies, and swimming performance experiments (Dudley and Platania 1997; Bestgen et al. 2010; Dudley et al. 2022). Swimming performance increases with body size and developmental stage, and therefore, habitat criteria for juvenile Rio Grande Silvery Minnows are reduced relative to adults. Criteria for larval Rio Grande Silvery Minnows were conservatively estimated using a general regression model relating total length (TL) to critical swimming speed for small freshwater fishes (≤ 60 mm TL) including cyprinids (Wolter and Arlinghaus 2004; Mortensen et al., 2019). These criteria were used to estimate the availability of physically suitable habitats from hydraulic modeling results.

The proposed hydraulic habitat criteria are approximate guidelines for providing physiologically suitable habitats throughout the life history of the Rio Grande Silvery Minnow, however, it is unrealistic to expect that these hydraulic criteria will represent all the necessary factors to ensure their survival. Developing stage-specific criteria involves setting fixed limits or boundaries on parameters that may not entirely reflect actual habitat occupancy or suitability across time and space. For example, adult Rio Grande Silvery Minnows may occupy areas with increased depths (>60 cm) if water velocity is sufficiently low (<40 cm/s; Dudley and Platania 1997).

Flow-Habitat Curves

Flow-habitat curves were generated using hydraulic modeling results and the hydraulic habitat criteria provided for the Rio Grande Silvery Minnow (Figure 19). Relationships between flow and habitat availability are widely used in environmental flow studies (e.g., Instream Flow Incremental Methodology) including previous studies of the Rio Grande Silvery Minnow (Bovee et al., 1998; 2008). For each subreach and discharge increment, hydraulic modeling results (HEC-RAS width-slices method) were post-processed using the hydraulic habitat criteria specified for the Rio Grande Silvery Minnow to calculate a metric of physical habitat availability for the primary life-stages of the species (i.e., the area within each subreach meeting the hydraulic habitat criteria was used to estimate the availability of physically suitable habitats). Specifically, the width-slices of each cross-section (i.e., agg/deg line) that met both the velocity and depth criteria were summed within each subreach and multiplied by 500 ft (approximate agg/deg line spacing) to estimate hydraulically suitable habitat areas. Habitat areas were normalized by the length of the respective subreach to account for variations in length (i.e., units of ft^2 per mile). Habitat availability was calculated incrementally across a range of discharges (500–10,000 cfs) to characterize the relationship between flow and habitat availability for each life-stage (larvae, juvenile, adult), subreach (e.g., B1–B4, Mo1–Mo5), and survey year (1992, 2002, 2012).

The relationships derived between flow and habitat availability for the Rio Grande Silvery Minnow are analogous to Weighted Usable Area in the Instream Flow Incremental Methodology (IFIM; Bovee et al., 1998). It is important to note that although habitat availability is calculated on an areal basis (i.e., area meeting habitat criteria), these quantities should be interpreted as indicators of physical habitat availability, not necessarily as precise quantifiers of habitat areas or numbers of fish (Reiser and Hilgert, 2018). The application and interpretation of the IFIM in studies of fish populations have received considerable attention (e.g., Reiser and Hilgert, 2018). Despite inherent limitations of such analyses, physical habitat modeling remains among the most widely applied and recognized analytical tools to assess flow-habitat relationships. Accordingly, the flow-habitat curves evaluated in this study were deemed a reasonable method to assess spatial and temporal variations in habitat availability for the Rio Grande Silvery Minnow. Overall, flow-habitat curves were used to evaluate how the availability of physically suitable habitats varies relative to discharge among life-stages, subreaches, and survey years.

Table 4. Hydraulically suitable habitat criteria (water velocity and depth) for principal life-stages of Rio Grande Silvery Minnow: larval, juvenile, and adult.

Life-stage	Depth, cm (ft)	Velocity, cm/s (ft/s)
Larval	0.1–15 (0.00–0.49)	0.0–5.0 (0.0–0.16)
Juvenile	1.0–50 (0.03–1.64)	0.0–30 (0.0–0.98)
Adult	5.0–60 (0.16–1.97)	0.0–40 (0.0–1.31)

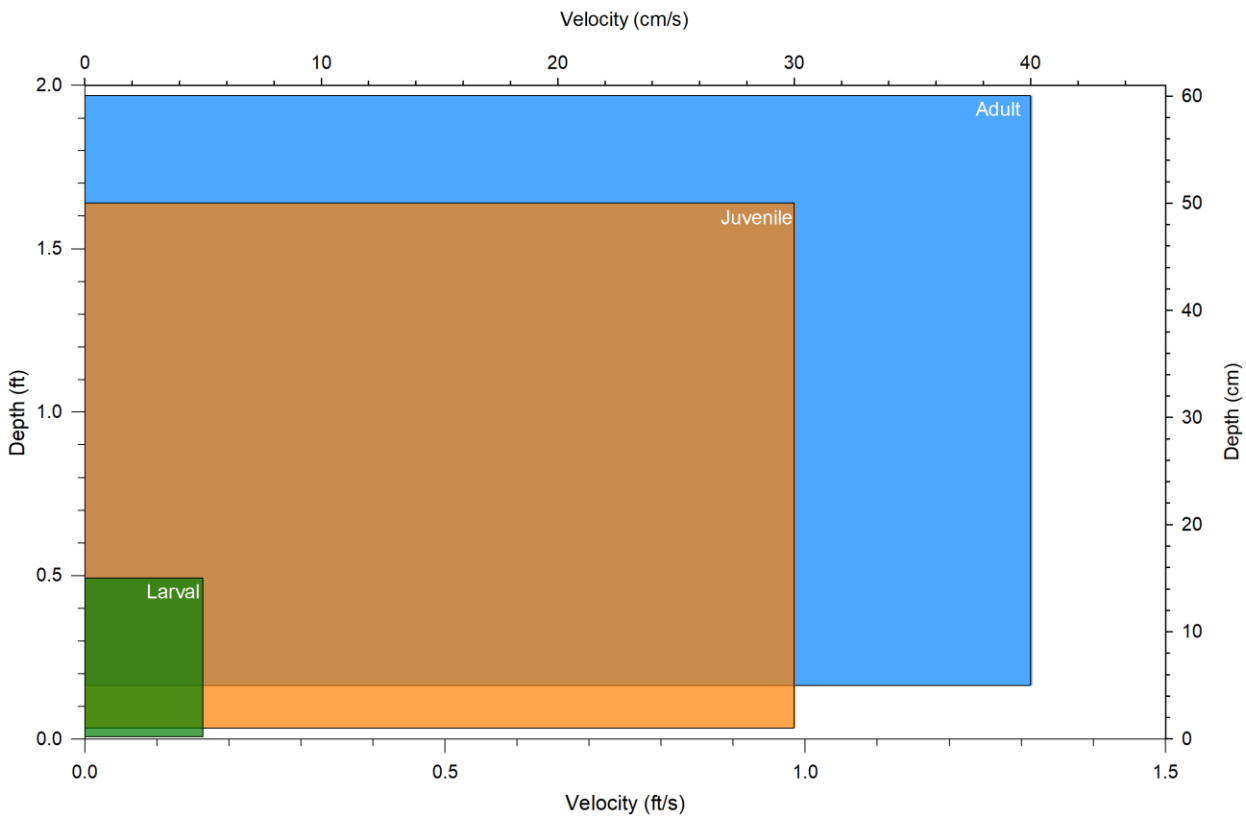


Figure 19. Hydraulically suitable habitat criteria (water velocity and depth) for principal life-stages of Rio Grande Silvery Minnow: adult, juvenile, and larval. Modified from Mortensen et al., 2019.

Time Integrated Habitat Metrics (1993–2021)

Habitat availability curves were integrated over time to calculate Time Integrated Habitat Metrics (TIHMs) for the Angostura Reach (Figure 20). This metric was developed to assess the interaction between discharge periods, habitat availability, and the population of the Rio Grande Silvery Minnow by providing a quantitative basis to compare the availability of physically suitable habitat areas over time to population parameters of the Rio Grande Silvery Minnow at the reach scale. Inputs required to calculate the TIHMs are daily discharge and flow-habitat curves for each decadal survey. Discharge data for the Angostura Reach were obtained from the following USGS gaging stations: the Rio Grande at Albuquerque, NM (USGS 08330000; 1993–2021). Habitat availability time series were calculated for each subreach (i.e., Bernalillo and Montaño). TIHMs were calculated corresponding to life-stage periods: larvae May–June, juveniles July–September, and adults October–April (Figure 20).

Conceptually, TIHMs represent the integral of habitat availability over time for each life-stage period in a given year as shown in eq. 1:

$$TIHM = \int_{t0}^{t1} H(t)dt \qquad (1)$$

Where H(t) is habitat availability as a function of time, t0 is the time at the beginning of the life-stage period, t1 is the time at the end of the life-stage period, and dt is differential time.

Habitat availability (ft^2/mi) was estimated at a daily time step (i.e., mean daily habitat availability). Mean daily habitat availability (H_t) was calculated by linear interpolation of flow-habitat curves between modeled flow profiles (e.g., between 1500 and 2000 cfs) as shown in eq. 2:

$$H(t) \cong H_t = mQ_t + b \qquad (2)$$

Where m is the slope between adjacent flow profiles, Q_t is the mean daily discharge on a given day during the life-stage period (t), and b is the intercept of the line between adjacent flow profiles (determined algebraically). The values for m and b varied between flow profiles for each life-stage and survey year.

Functionally, the TIHM was approximated by a finite sum of mean daily habitat availability values (e.g., Riemann sum) over each life-stage period as shown in eq. 3:

$$TIHM \cong \sum_{T=t0}^{t1} H_t \cdot \Delta T \qquad (3)$$

Where ΔT = 1 day. Units of the habitat metrics are: TIHM (ft^2 day/mi) = H_t (ft^2/mi) •ΔT (day). Non-normalized spatial values (e.g., acres) can be calculated as:
Acres = TIHM (ft^2 day/mi) • Length (mi) / Duration (t_1–t_0; days) • unit conversion (1 acre / 43,560 ft^2). Although spatial units can be derived, TIHM values should be interpreted as an index of habitat availability and are not intended to be a precise quantifier of habitat area (Reiser and Hilgert, 2018).

TIHMs were calculated for the period corresponding to population monitoring data for the Rio Grande Silvery Minnow (1993–2021) to serve as a covariate for analysis of long-term ecological relationships. For interim years between survey periods (i.e., 1992, 2002, 2012), the preceding hydraulic model was applied (e.g., TIHMs 1993–2001 were calculated from the 1992 hydraulic model). This methodology was deemed reasonable based on relatively low magnitude change in flow-habitat relationships 1992–2012.

TIHMs were also used to evaluate the effects of geomorphic change on habitat availability using historical channel geometries. Due to the strong influence of hydrology on habitat availability results (i.e., TIHMs), hydrologic conditions were isolated to assess the relative influence of geomorphic changes on the TIHMs over time. Annual hydrographs (by water year) were selected to represent three flow scenarios within the study period: low flow (2003, 2012), moderate flow (2004, 2007), and high flow (1994, 2005) – discharge data were obtained from USGS 08330000 (Rio Grande at Albuquerque, NM). TIHMs for each life-stage period (described above) were calculated for each of these hydrographs using available channel geometries (1962, 1972, 1992, 2002, 2012). For this analysis, TIHMs were also calculated at the subreach scale to enable assessment of the effects of geomorphic change across space and time; discharge was held constant across subreaches to provide a basis for equivalent comparisons.

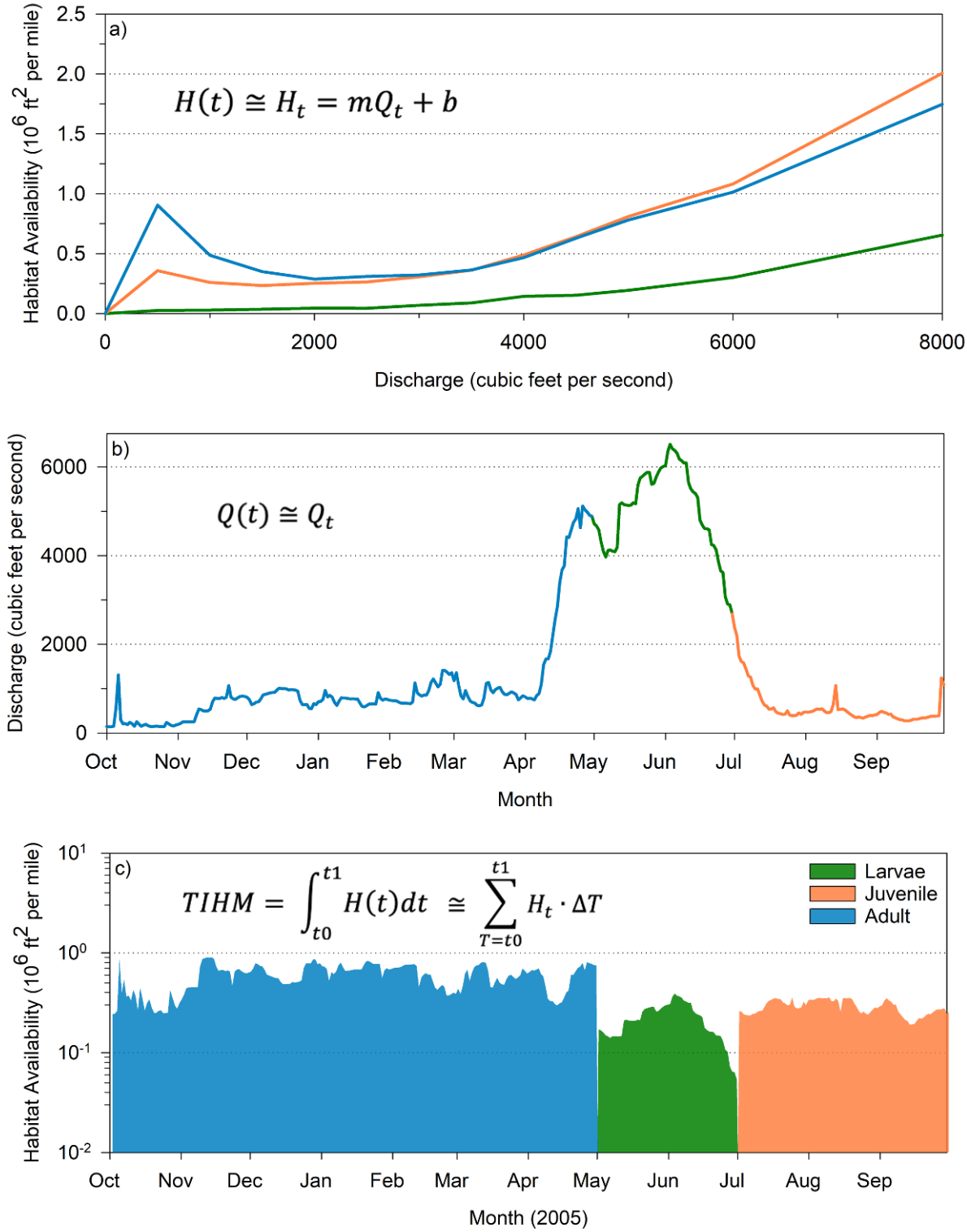


Figure 20. Example (a) flow-habitat curves, (b) hydrograph, and (c) habitat availability time series for the Angostura Reach during WY 2005. Shaded areas in the bottom panel represent TIHMs by life-stage (i.e., area under the curve, integral). Note log-scale in bottom panel.

Channel-Habitat Evolution Models

Geomorphic data and flow-habitat curves were aggregated graphically to assess relationships between channel evolution and habitat availability over time. Stages of the planform evolution model for the Middle Rio Grande (Massong et al., 2010) were used to describe distinct changes in channel morphology in the Angostura Reach. Stages of this geomorphic model were assigned for each subreach and decadal survey 1962–2012. Stages of the planform evolution model were further evaluated using representative cross-sections, aerial imagery, and flow-habitat curves. Cross-section data were used to illustrate the magnitude and rate of channel incision and narrowing over time. Aerial imagery was used to show characteristic river planforms over time and morphological trends for each subreach. Flow-habitat curves were used to assess interactions between discharge, channel morphology, and habitat availability, and how these relationships have changed through time and space. Changes in channel morphology were also considered with respect to established and recently developed models of channel evolution (e.g., Castro and Thorne, 2019; Booth and Fischenich, 2015; Cluer and Thorne, 2013; Schumm et al., 1984; Simon and Hupp, 1986; Rozin and Schick, 1996). The channel-habitat evolution model developed herein was used to create a more comprehensive view of channel evolution in the Middle Rio Grande and the subsequent impacts such processes have on the physical habitat conditions required by the Rio Grande Silvery Minnow.

Long-Term Ecological Relationships

Ecological relationships between environmental conditions and the population of Rio Grande Silvery Minnow were evaluated for the Angostura Reach over 28 years. Covariates considered for modeling sampling-site density data included TIHMs, corresponding to life-stage periods, and various flow metrics. Flow metrics were calculated for life-stage periods (Larval: May–June [31 days], Juvenile: July–September [61 days], and Adult: October–April [213 days]); the same periods that were used to calculate TIHMs. Month ranges coincided with the highest prevalence of each life-stage in the wild. It is noted that overlap among life-stages occurs, particularly for adults (i.e., adult fish are present May–September), however, the short lifespan of the species (1–2 yrs) prompted selection of periods corresponding to the highest prevalence of life-stages, as these periods were suspected to correspond to recruitment and survival of the respective life-stages. Additional flow metrics were included to characterize spring runoff and low flow conditions during life-stage periods (described below). These covariates were selected to account for temporal variation in flow and habitat availability during the study period. Assessing relationships among environmental conditions and the occurrence/density of the Rio Grande Silvery Minnow can indicate underlying ecological processes that drive population responses over time.

Statistical Analyses

Robust ecological modeling approaches were implemented to quantitatively evaluate the effects of environmental variables on long-term trends in the occurrence and density of the Rio Grande Silvery Minnow (Dudley et al., 2022). Population parameters derived for analyses included estimated density, $E(x)$, occurrence probability, δ, and lognormal density, μ. Rio Grande Silvery Minnow sampling-site density data during October (1993–2021), based on small-mesh regular seine samples, were analyzed using PROC NLMIXED (Nonlinear Mixed Models; SAS, 2023). This advanced numerical optimization procedure was used to fit our long-term data to a mixture model, which comprised a binomial distribution (i.e., based on presence-absence data) and a lognormal distribution (i.e., based on natural logarithms of nonzero data). Mixture models (e.g., combining a binomial distribution with a lognormal distribution) are particularly effective for modeling zero-inflated data (White, 1978; Welsh et al., 1996; Fletcher et al., 2005; Martin et al., 2005) and for evaluating the effects of environmental covariates on population parameters. Logistic regression was used to estimate the annual probability that a site was occupied (i.e., occurrence probability), and a lognormal model was used to estimate the annual lognormal density based on occupied sites. Numerical optimization of the models provided four estimates (estimated occurrence probability [δ], estimated lognormal density [μ], standard deviation of the estimated lognormal density [σ],

and estimated density [$E(x)$, based on δ, μ, and σ]) for each year (i.e., based on the site-specific sampling data). Values of $E(x)$ could not be estimated, however, when only a single nonzero value was recorded (i.e., precluding mixture-model estimation of σ). Naïve density estimates (i.e., unmodeled), calculated using the method of moments (Zar, 2010), were added as a reference to applicable figures.

Ecological relationships, interactions between environmental parameters (e.g., TIHMs and flow metrics) and the fish population (i.e., E(x), δ, μ) were evaluated using generalized linear models. Generalized linear models were based on environmental covariates (i.e., independent variables), TIHMs and flow metrics, and population parameter estimates (δ, μ, and σ [i.e., dependent variables]), where a logit link was used for δ, an identity link for μ, and a log link for σ. The logit link maintains δ on a 0–1 scale, the identity link maintains μ between –infinity and +infinity, and the log link maintains σ greater than zero. In the simplest case with no covariates and no random effects, the mixture-model structure can be considered a zero-inflated lognormal model for estimated densities. In all analyses, a categorical covariate for sampling year (Year) was included to represent the maximum variation attributable to time effects. As no other time-effects model can explain all the variation, the year (or global) model (δ[Year] μ[Year]) represents the upper limit on the amount of explainable variation and the null model (δ[.] μ[.]) represents the lower limit of that variation. Additionally, all nested environmental covariates varied across Year and were assessed for their effectiveness in explaining the total time-specific variation of the population parameters (i.e., ecological models).

Environmental covariates considered for modeling October sampling-site density data (1993–2021) included various metrics based on habitat and flow data. Habitat availability, based on the Time Integrated Habitat Metric (TIHM: 10^6 ft^2 day/mi), was estimated annually for larvae (MayJunHab), juveniles (JulSepHab), and adults (OctAprHab). For example, the most recent TIHMs (2021) were calculated for larval habitat (May to June 2021; 61 days), juvenile habitat (July to September 2021; 92 days), and adult habitat (October 2020 to April 2021; 212 days). TIHMs were log-transformed (based on the natural logarithm) prior to analysis, as TIHMs for all life-stages were found to increase exponentially as a function of discharge across the range of flows observed during this study (i.e., log-transformation better facilitated habitat/flow comparisons). The first set of flow metrics was based on mean daily discharges (cfs) for larvae (MayJunMean), juveniles (JulSepMean), and adults (OctAprMean). The second set of flow metrics was based on high flows for larvae (MayJun28dHigh), and low flows for juveniles (JulSep7dLow) and adults (OctApr7dLow). For example, the most recent values were based on the highest 28 days (one month) of flow for larvae (May to June 2021), and the lowest 7 days (one week) of flow for juveniles (July to September 2021) and adults (October 2020 to April 2021). The flow duration for larvae was based on the approximate time required for eggs to develop beyond the vulnerable early larval phases (i.e., protolarvae and mesolarvae; Platania, 1995b). The flow duration for juveniles and adults was based on the approximate time required for low flows to negatively affect fish throughout the reach (i.e., based on declining flows, isolated pools, and river drying; Cave and Smith, 1999; Archdeacon, 2016). Fixed-effects models for each covariate were generalized linear models with the corresponding link function. These fixed effects assume that variation in the dataset is explained by the covariate. For δ, there is no over-dispersion or extra-binomial variation, and for μ, no extra variation provided beyond the constant σ model. Random-effects models (R) were also considered for δ and μ to provide additional variation around the fitted line where a normally distributed random error with mean zero, and nonzero standard deviation, was used to explain deviations around the fitted covariates. All random effects were integrated out of the likelihood (see Pinheiro and Bates, 1995) during model fitting.

Goodness-of-fit statistics (logLike = –2[log-likelihood] and AIC_c = Akaike's information criterion [Akaike 1973; Burnham and Anderson, 2002] for finite sample sizes) were generated to assess the relative fit of data to various mixture models. Lower values of AIC_c indicate a better fit of the data to the model. Models were ranked by AIC_c values, and the top ten models (based on AIC_c weight [w_i]) were presented. As nested environmental covariates were only used individually to model the population parameters (i.e., no additive effects), potential issues of multicollinearity were avoided. Further, AIC_c model selection ranks single-variable models appropriately, even if variables are highly correlated (i.e., resulting w_i values would be similar). An analysis of deviance (ANODEV) was used to determine the relative proportion of deviance in logLike values explained by the environmental covariates, for both δ and μ models, and to assess whether that proportion was significantly different from zero ($P < 0.05$) based on an F-test (Skalski et al., 1993). Detailed statistical methods and assumptions are presented in the Rio Grande Silvery Minnow Population Monitoring Program reports (e.g., Dudley et al., 2022).

Process-Linkage Framework

The determination of linkages among fluvial and ecological processes across the multiple spatial and temporal scales at which they operate is inherently complex, and therefore, can be aided by conceptual models and frameworks. Conceptual hierarchical models have been used to illustrate and describe the multi-scale interactions among watershed inputs, geomorphic processes and attributes, habitat conditions, and biotic responses, including impacts by natural and anthropogenic factors in several large river systems (Jacobson et al., 2014; Trinity River Restoration Program, 2009; Stillwater Sciences, 2007). Accordingly, a simplified conceptual model was developed to represent the Middle Rio Grande, including specific geomorphic processes that are suspected to influence the geomorphic attributes and habitat conditions required by the Rio Grande Silvery Minnow (Figure 21). In this model, watershed inputs (primarily water and sediment) drive the processes that determine channel and floodplain morphology and subsequently, the habitat conditions of the river-floodplain system. As such, anthropogenic activities or natural factors that alter inputs, processes, or geomorphic attributes, will in turn impact habitat conditions and biotic responses. For example, reduction in spring runoff events (inputs) can alter the timing, frequency, and duration of floodplain inundation (processes), which can modify channel and floodplain morphology (geomorphic attributes). Further, changes in inundation patterns can affect the seasonal availability of floodplain habitats (habitat conditions) and impact the Rio Grande Silvery Minnow population dynamics (biotic response). Additionally, feedbacks can occur among processes and geomorphic attributes, such as, riparian colonization of floodplain and channel surfaces increases roughness, increasing sediment deposition and floodplain accretion during flooding, altering channel-floodplain morphology, and ultimately reducing the frequency and extent of floodplain inundation over time. This model was developed as a tool to refine our conceptual understanding of the complex dynamics of the Middle Rio Grande ecosystem, identify dominant linkages among fluvial geomorphic processes and habitat conditions required by the Rio Grande Silvery Minnow, and stimulate hypotheses related to key process-linkages.

The hierarchical model was further developed to provide a framework to investigate process-linkages. The colored boxes in Figure 21 represent sub-models that illustrate how linkages are characterized and assessed using available data, analytical methods, and current knowledge of the ecosystem. The three stages provided by the REFORM framework were incorporated into this approach: (1) delineation of spatial units, (2) characterization of spatial units using existing data sets, and (3) assessment of past and present river characteristics (Gurnell et al., 2016). Figure 22 (brown box in Figure 21) delineates the spatial units of this study, including the multi-scale interactions among watershed inputs, fluvial geomorphic processes, and geomorphic attributes, and long-term systematic data collection efforts in the Middle Rio Grande (e.g., agg/deg lines, RGSM monitoring sites, gaging stations). Figure 23 (blue box) shows how dynamic interactions between geomorphic attributes and habitat conditions (at reach and subreach units) are characterized and assessed using existing datasets (e.g., agg/deg lines) and appropriate methods (e.g., hydraulic modeling, habitat suitability). In Figure 23 (green box), habitat conditions, environmental factors (e.g., streamflow), and knowledge of the ecology of the Rio Grande Silvery Minnow are integrated to assess how these factors influence population dynamics of the species. Specific methods are described in the following section. Altogether, these models form a conceptual framework to investigate process-linkages in the Middle Rio Grande.

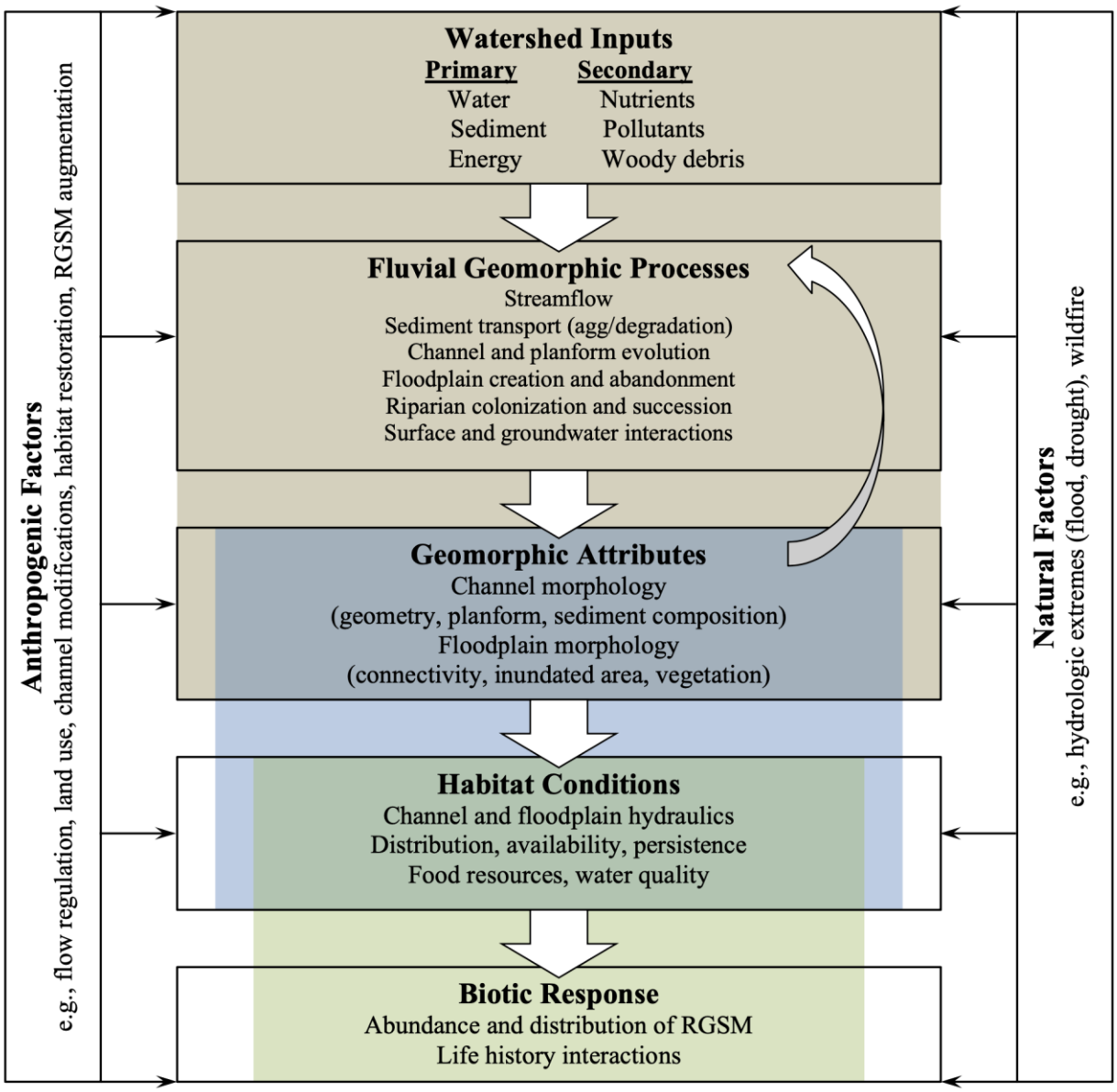


Figure 21. Simplified conceptual model of the linkages between watershed inputs, fluvial geomorphic processes and attributes, habitat conditions, and the biota of the Middle Rio Grande ecosystem. Colored boxes represent nested sub-models. Modified from Stillwater Sciences et al. (2007) and Trinity River Restoration Program (2009).

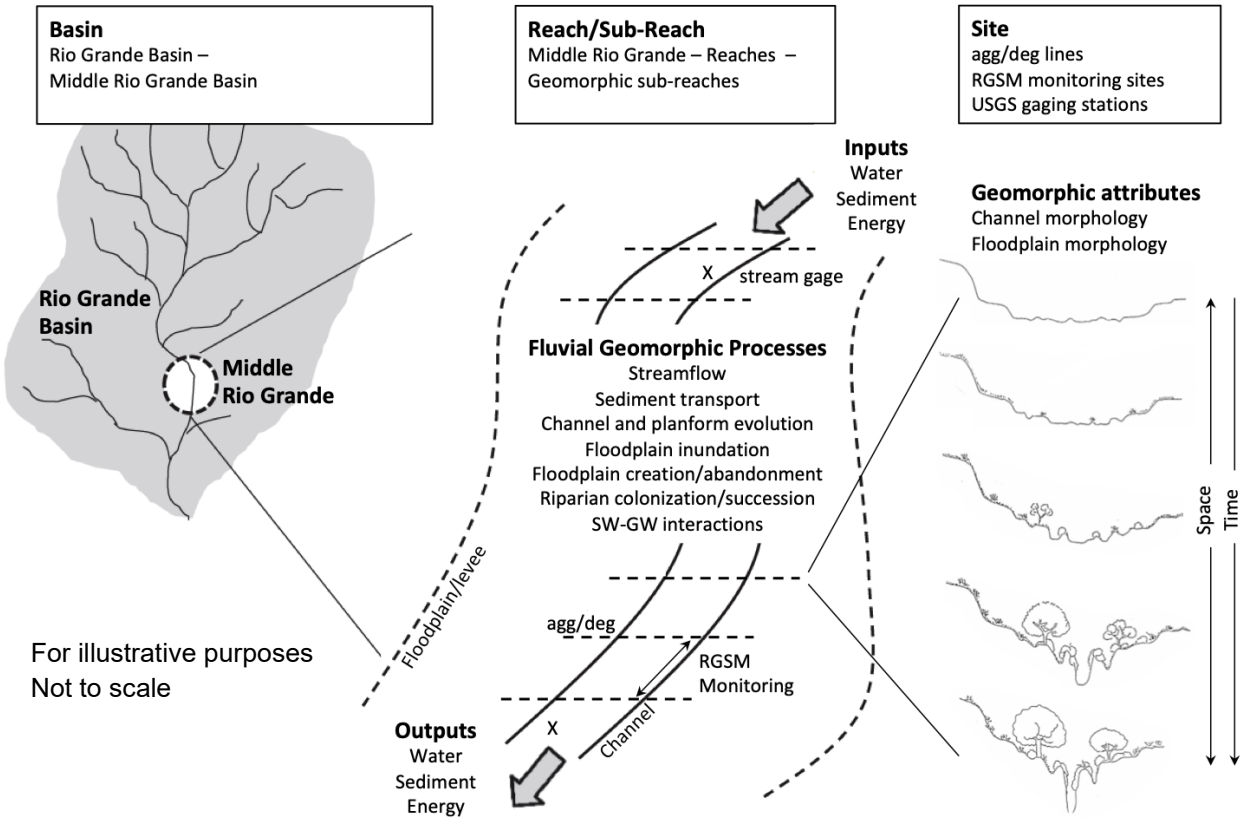


Figure 22. Delineation of spatial units used in this study, including the interactions among watershed inputs, fluvial geomorphic processes, and geomorphic attributes, and long-term systematic data collection efforts in the Middle Rio Grande (e.g., agg/deg lines, RGSM monitoring sites, stream gages). Basin and Reach panels modified from Wohl et al. (2015); illustrated channel cross-sections are from Rozin and Schick (1996).

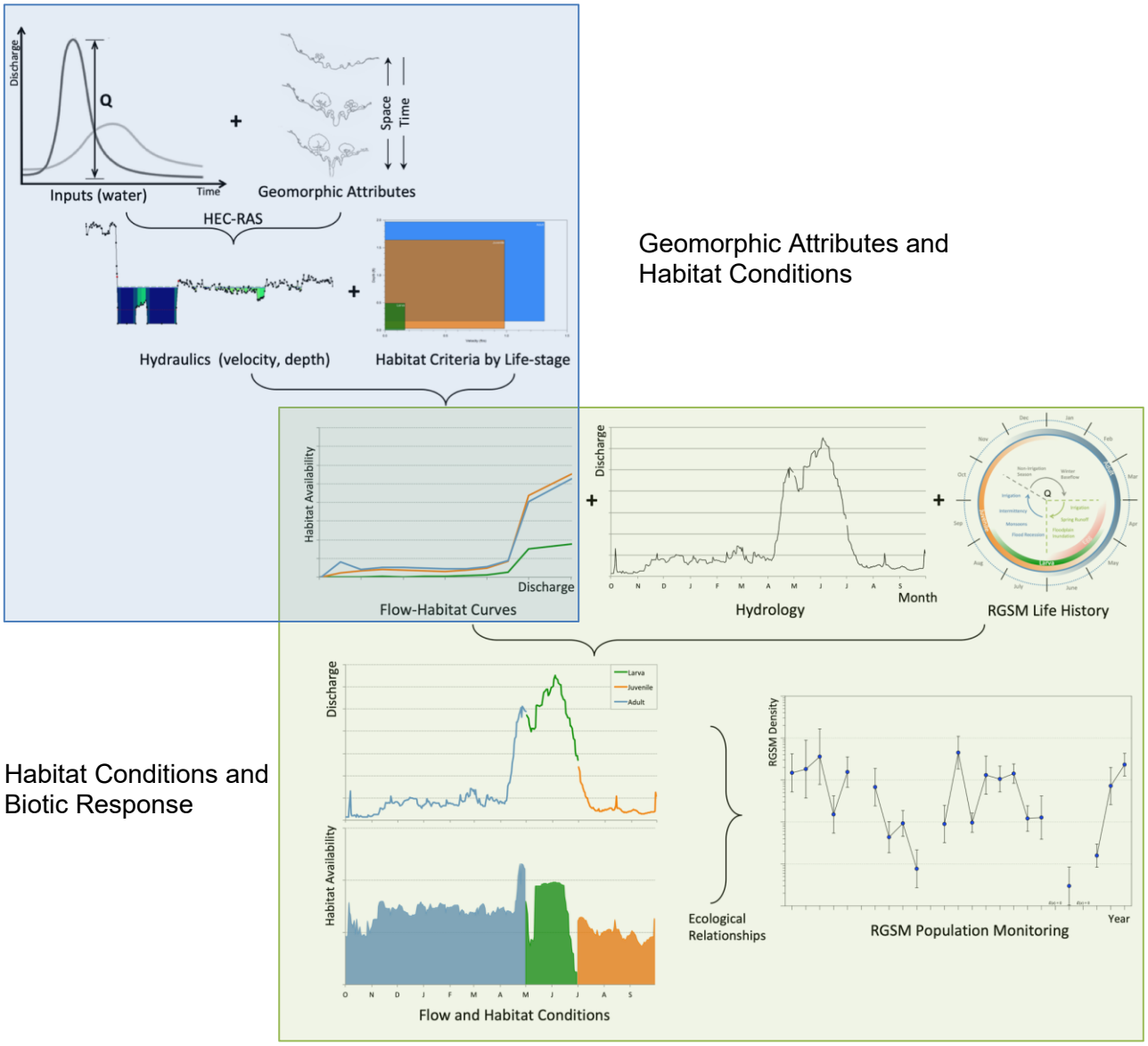


Figure 23. Characterization and assessment of geomorphic attributes, hydraulics, and physical habitat conditions through time including their interaction with environmental factors and the life history of the RGSM. Colored boxes (blue, green) correspond to Figure 21.

RESULTS

This section presents the primary results obtained for the Angostura Reach. A subset of results was selected for inclusion in this section; supplementary results are provided in Appendix A.

Reach and Habitat Analyses – Angostura Reach

Flow-Habitat Curves

The width-slices method was used to determine quantitative relationships between habitat availability and discharge. Habitat availability for the primary life-stages of the Rio Grande Silvery Minnow (larva, juvenile, and adult) was calculated at 500 cfs intervals between 500–10,000 cfs to characterize flow-habitat relationships for survey years 1962–2012 (Figure 24). Flow-habitat curves were assumed to be constant for years between surveys 1992–2012 – flow-habitat curves were not adjusted for temporal variation between survey periods due to low magnitude of change during this period. Curves were obtained for subreach-scale and reach-scale (Figure 24).

Flow-habitat curves were calculated for the primary life-stages of the Rio Grande Silvery Minnow using hydraulically suitable habitat criteria (Mortensen et al., 2019). Curves for juvenile and adult life-stages were similar in magnitudes and responses to discharge. Larval habitat availability was lowest in magnitude of the principal life-stages and showed only marginal increases with increasing discharge 1992–2012.

Habitat availability curves varied temporally during the study period and spatial variation was relatively minor (Figure 24). Flow-habitat curves in the Angostura Reach were largely characterized by two periods: 1962–1972 and 1992–2012. Curves 1962–1972 showed large, gradual increases in habitat availability with increasing discharges with peaks at approximately 2,000 cfs for larval criteria and 4,000–6,000 cfs for juvenile and adult; peak habitat availability for this period was $1.0 \cdot 10^6$ ft^2 day per mile for the larval life-stage, and $4.2–4.4 \cdot 10^6$ ft^2 day per mile for juvenile and adult life-stages, respectively. Curves 1992–2012 showed muted responses to discharges <4,000 cfs with a relatively low magnitude peak occurring at 500 cfs and larger increases with discharges >4,000 cfs. Peak habitat availability substantially decreased 1972–2012; in 2012 peak habitat availability (recorded at 8,000 cfs) was $0.21 \cdot 10^6$, $2.1 \cdot 10^6$, and $2.3 \cdot 10^6$ ft^2 day per mile for larval, juvenile, and adult life-stages, respectively. The shape of the flow-habitat curves (e.g., inflection points) suggest very low overbanking discharges in 1962–1972 with a large increase in bankfull discharge to around 4,000 cfs occurring between 1972 and 1992. Differences between Bernalillo and Montaño subreaches were generally minimal, however, for years 1992–2012 Bernalillo showed slightly higher levels of habitat availability for discharges <5,000 cfs.

Habitat Mapping (2012)

Habitat maps were generated for three discharges (1,500, 3,000, and 5,000 cfs) to illustrate the locations of hydraulically suitable habitats (by life-stage) obtained from the hydraulic model. Results showed the formation of habitats was concentrated in inundated secondary channels in the Bernalillo subreach, and to a lesser extent in the Montaño subreach. A subset of habitat maps is included to illustrate conditions representative of each subreach (Figures 25–26); the complete set of habitat maps are presented for the Bernalillo subreach in Radobenko et al., (2023) and for the Montaño subreach in Anderson et al., (2023).

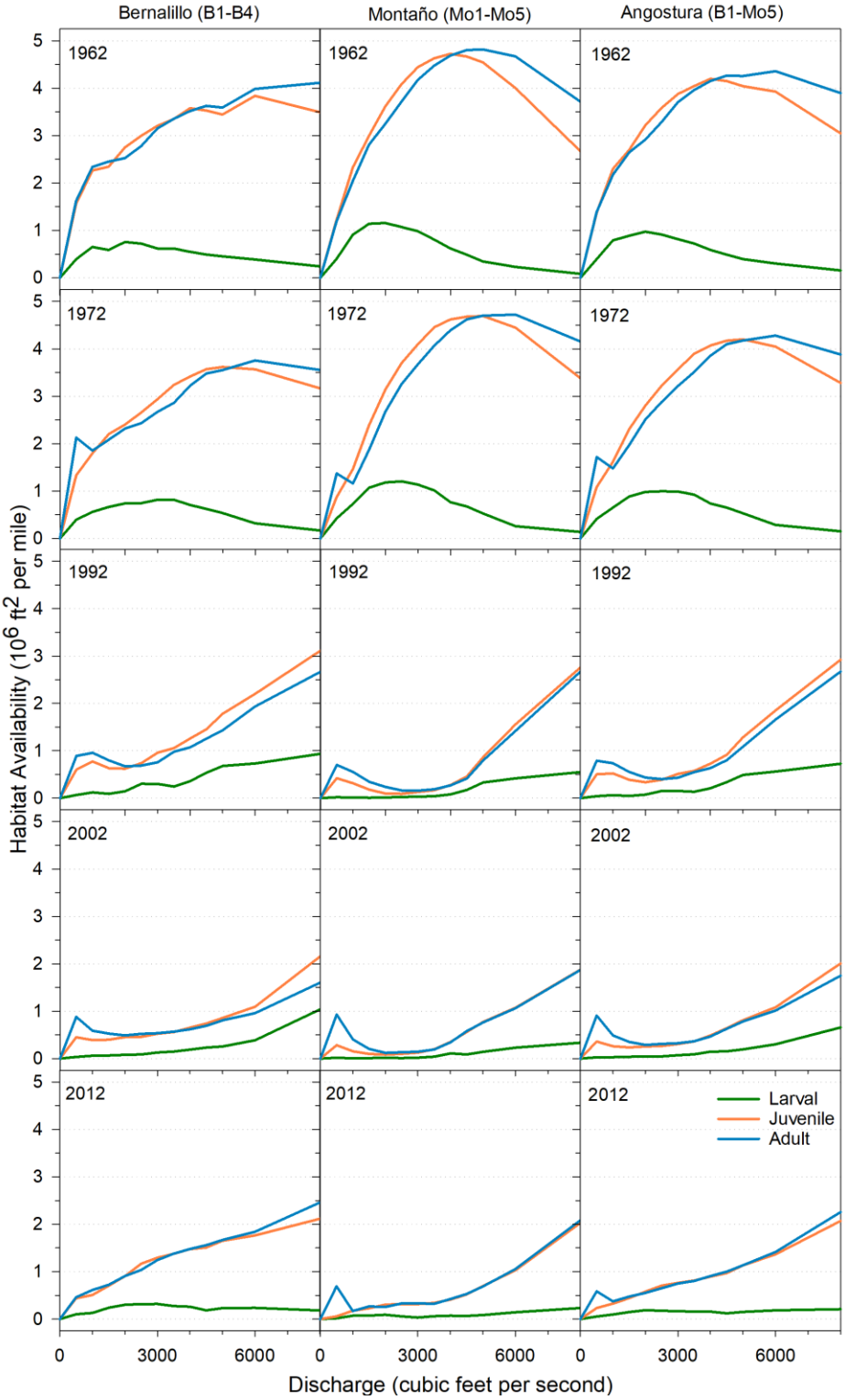


Figure 24. Flow-habitat curves for the Angostura Reach. Columns left to right are Bernalillo and Montaño subreaches and the combined Angostura Reach. Curves are shown through time top to bottom (1962–2012). Line colors represent the primary life-stages of the Rio Grande Silvery Minnow. Habitat availability was normalized by reach length.

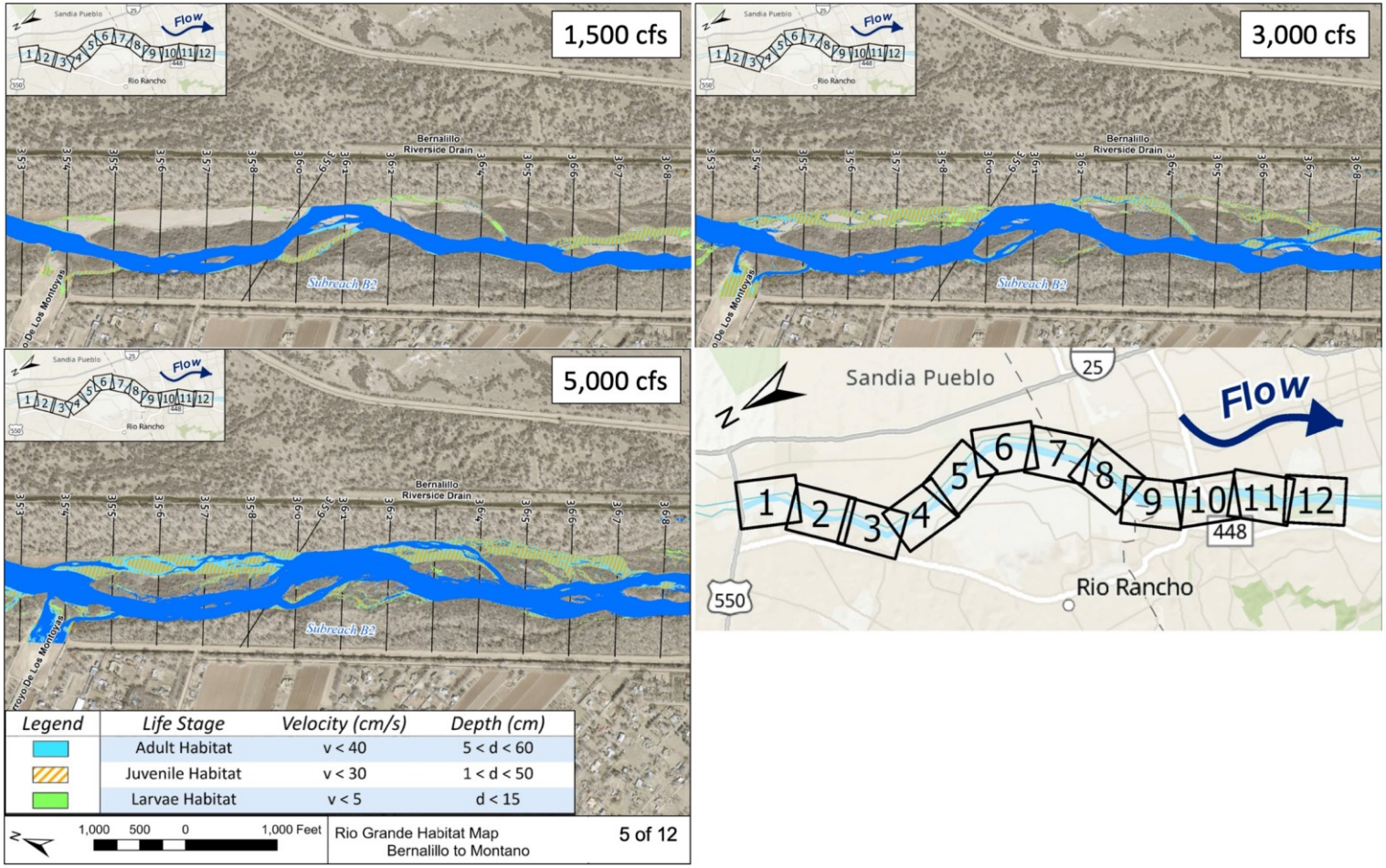


Figure 25. Habitat mapping results from subreach B2 (2012) showing locations of hydraulically suitable habitats by life-stage across three discharges: 1,500, 3,000, and 5,000 cfs. Map 5 of 12 (inset map). From Radobenko et al., (2023).

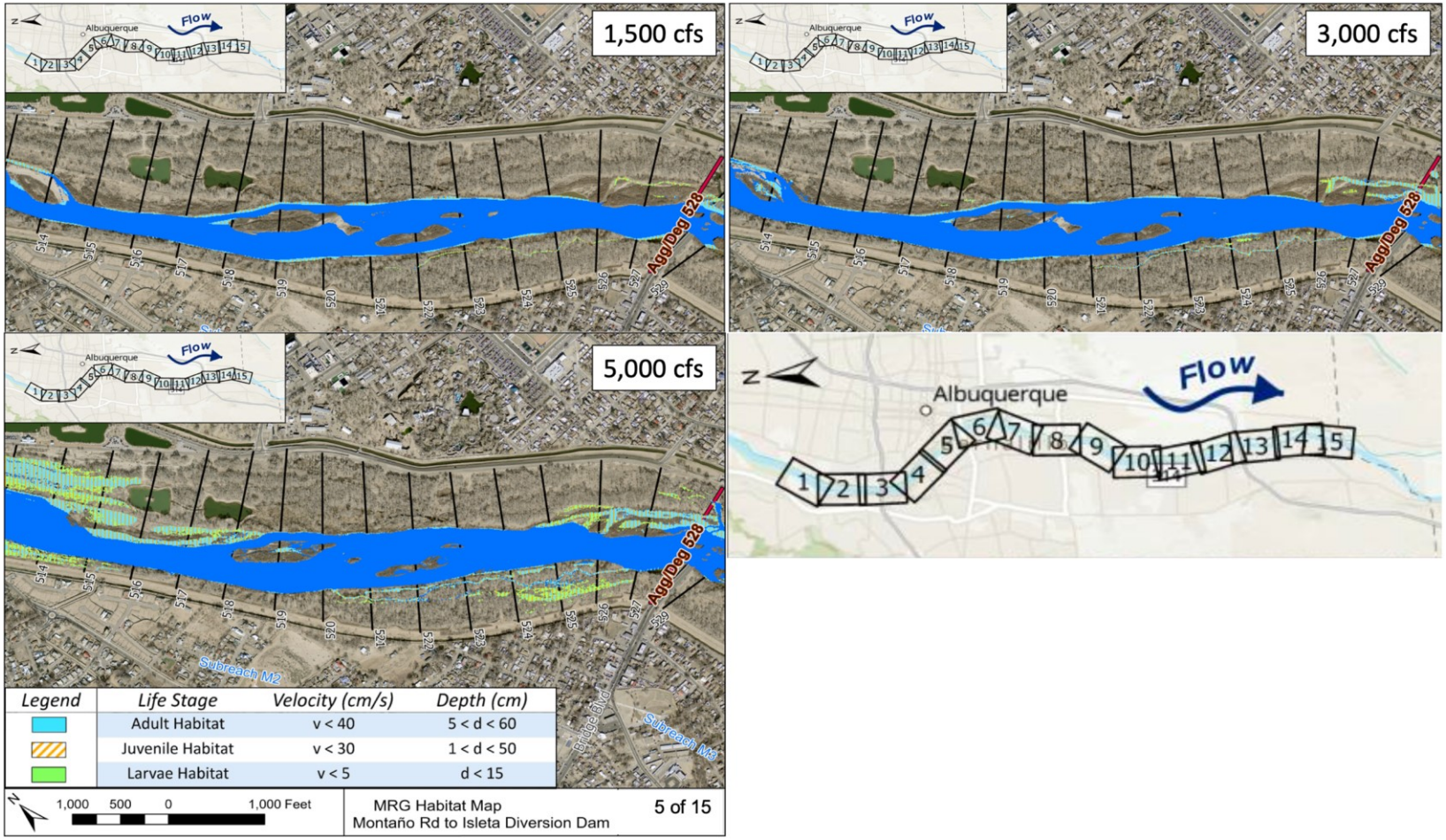


Figure 26. Habitat mapping results for subreach Mo2 (2012) showing locations of hydraulically suitable habitats by life-stage across three discharges: 1,500, 3,000, and 5,000 cfs. Map 5 of 15 (inset map). From Anderson et al., (2023).

Time Integrated Habitat Metrics (TIHMs)

Time Integrated Habitat Metrics (TIHMs) were calculated annually 1993–2021 for seasonal periods corresponding to the primary life-stages of the Rio Grande Silvery Minnow. Flow-habitat curves (Figure 24), Angostura Reach discharge values (Figure 27), and life-stage periods were used to calculate TIHMs. A total of 87 TIHMs were calculated for the study period (Table 5).

Time Integrated Habitat Metrics varied through time and by life-stage period (Table 5, Figures 28–29). Larval TIHMs ranged from 1.59–27.76 10^6 ft^2 day/mi, juvenile TIHMs ranged 7.41–67.86 10^6 ft^2 day/mi, and adult TIHMs ranged 87.14–167.85 10^6 ft^2 day/mi. Mean TIHMs for the study period were lowest for the larval life-stage (7.80 ± 7.34 10^6 ft^2 day/mi) and were followed by the juvenile life-stage (29.57 ± 14.01 10^6 ft^2 day/mi). Adult TIHMs were the highest on average (129.11 ± 24.87 10^6 ft^2 day/mi) and had the highest variance of the life-stage periods. Median TIHMs were highest for adults (135.92 10^6 ft^2 day/mi), followed by juveniles (27.71 10^6 ft^2 day/mi), and were lowest for larvae (5.40 10^6 ft^2 day/mi). Subreach contributions to TIHMs varied across years and life-stages. For larval TIHMs, the Bernalillo subreach contributed the most across years with a range 64–88% of the total. For juvenile TIHMs, the Bernalillo subreach contributed approximately 60% of the total 1993–2011; 2012–2021 showed percent contribution by this subreach increased to approximately 80%. Adult TIHMs showed near equal contributions by the two subreaches across years. Overall, larval TIHMs tended to be the lowest in magnitude and adult TIHMs tended to be the highest, yet all life-stages showed considerable variation through time, primarily related to variations in hydrologic conditions.

TIHMs captured interactions between channel morphology and hydrology, specifically seasonal flow conditions corresponding to each life-stage period (Figures 30). Larval TIHMs were sensitive to the magnitude and duration of peak flows May–June and juvenile TIHMs were sensitive to the duration and frequency of low flows July–September. TIHMs for adults were less affected by discharge fluctuations due to the tendency for relatively stable conditions to occur October–April across years.

Effects of Geomorphic Changes on TIHMs

The effects of temporal geomorphic changes on habitat metrics (TIHMs) 1962–2012 were assessed using selected annual hydrographs (Figure 31). Due to the influence of annual hydrology on the TIHMs (Table 5, Figure 30), six annual hydrographs representing three flow scenarios (i.e., low, moderate, and high) were selected to assess the relative influence of geomorphic changes on TIHMs. Discharge data were obtained from USGS 08330000 (Rio Grande at Albuquerque, NM) and held constant for the reach for this analysis.

Overall, Time Integrated Habitat Metrics decreased between 1962 and 2012 for each of the flow scenarios investigated (Table 6; Figures 32–34. Interim survey years (1972–2002) showed variable trends between 1962 and 2012, however, an overall declining trend is evident for nearly all flow scenarios during the study period 1962–2012. For the larval life-stage, the low and moderate flow scenarios showed large declines in TIHMs 1972–1992 and remained relatively stable 1992–2012; the high flow scenarios showed modest declines over time. For all flow scenarios, the juvenile life-stage, TIHMs showed progressive declines each survey 1962–2012. Adult TIHMs showed considerable declines 1972–1992 and remained relatively stable 1992–2012 for all flow scenarios.

Between 1962 and 2012, decreases to TIHMs varied by flow scenario and life-stage period. On average, the greatest magnitude of change 1962–2012 for larval TIHMs was for low flow scenarios (87%; 2003 and 2012), followed by moderate flow scenarios (83%; 2004 and 2007), and least for high flow scenarios (66%, 1994 and 2005). For juvenile TIHMs, decreases were relatively constant across flow scenarios – TIHMs on average decreased by 83–84% between 1962 and 2012. Declines in adult TIHMs between 1962 and 2012 were on average greatest for moderate and high flow scenarios (72–79%) and lowest for the low flow scenarios (64%).

Subreach contributions to TIHMs varied by life-stage and flow scenario. Larval TIHMs showed a notable increase in percent contribution from the Bernalillo subreach 1972–1992; differences in subreach contributions were less pronounced for high flow scenarios. Juvenile TIHMs showed relatively constant and equal subreach contributions 1962–2002 with a considerable increase in percent contribution by Bernalillo in 2012 for all flow scenarios. Adult TIHMs showed relatively constant and equal subreach contributions 1962–2012 for all flow scenarios.

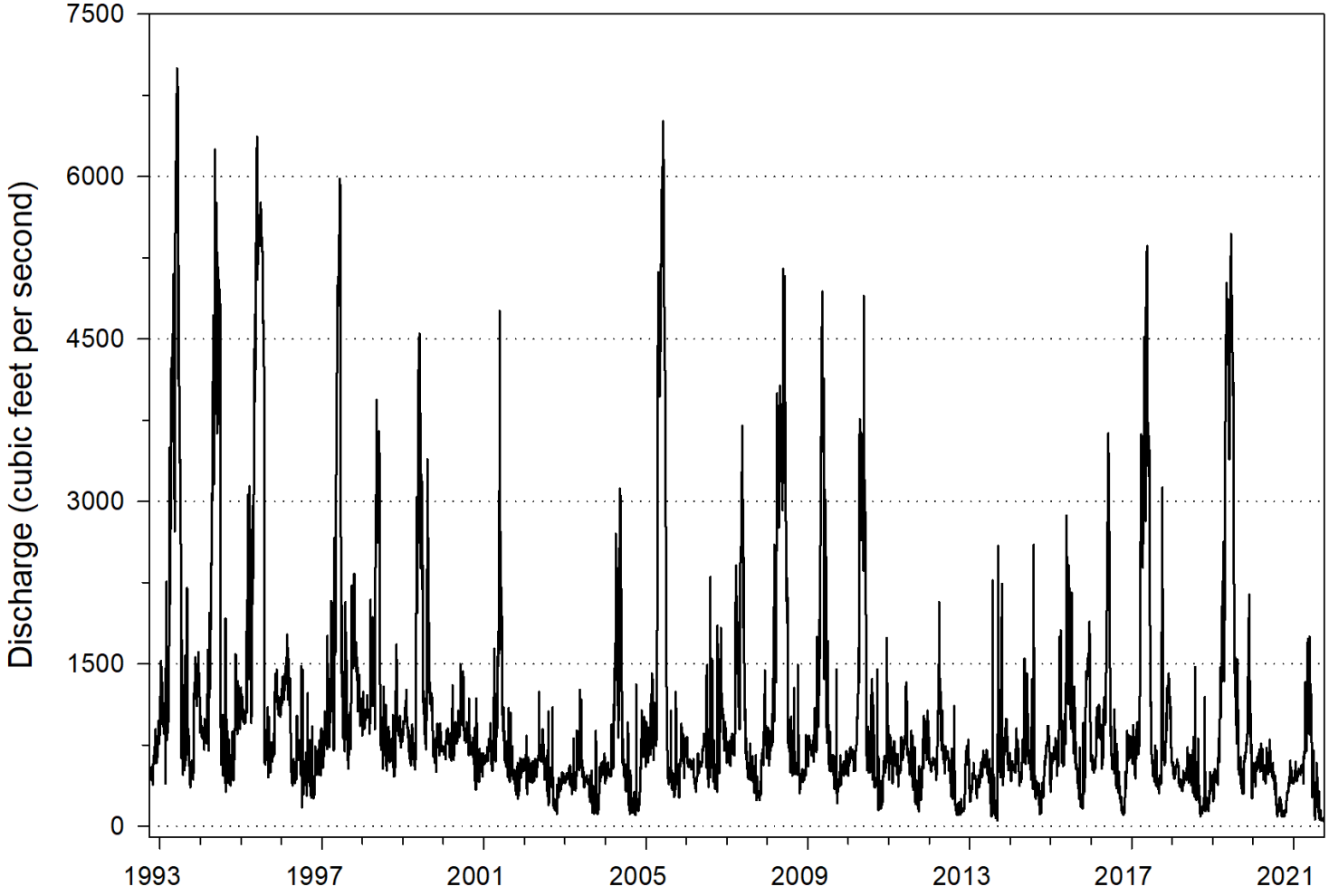


Figure 27. Hydrograph (mean daily discharge) for the Angostura Reach during the study period (WY 1993–2021). Discharge data were obtained from USGS 08330000 (Rio Grande at Albuquerque, NM).

Table 5. Time Integrated Habitat Metrics (TIHMs) by life-stage period for the Angostura Reach 1993–2021.

	TIHMs (10^6 ft^2 day per mile)		
Water Year	Larval[1]	Juvenile[2]	Adult[3]
1993⁻	26.00	45.06	146.31
1994⁻	23.84	43.55	140.12
1995⁻	27.76	67.86	135.92
1996⁻	2.73	39.08	142.75
1997⁻	18.84	44.15	145.97
1998⁻	6.71	46.99	134.94
1999⁻	10.63	42.90	157.80
2000⁻	3.49	46.76	161.53
2001⁻	5.65	45.55	158.56
2002^	1.67	25.79	167.85
2003^	1.59	21.69	148.05
2004^	2.25	16.72	139.82
2005^	13.26	26.51	123.23
2006^	1.68	27.71	163.13
2007^	2.79	28.60	123.43
2008^	6.55	29.41	117.62
2009^	5.39	28.93	139.90
2010^	3.81	28.35	152.12
2011^	1.75	25.25	149.20
2012⁺	4.08	12.75	100.07
2013⁺	3.69	15.85	87.14
2014⁺	5.40	18.02	109.64
2015⁺	7.96	24.73	100.24
2016⁺	8.52	19.22	95.77
2017⁺	8.83	19.77	103.81
2018⁺	3.19	14.30	105.01
2019⁺	8.69	35.06	89.29
2020⁺	3.51	9.66	108.56
2021⁺	6.02	7.41	96.42

[1] = Larval life-stage corresponds to May–June
[2] = Juvenile life-stage corresponds to July–September
[3] = Adult life-stage corresponds to October–April
⁻ = Derived from 1992 channel geometry
^ = Derived from 2002 channel geometry
⁺ = Derived from 2012 channel geometry

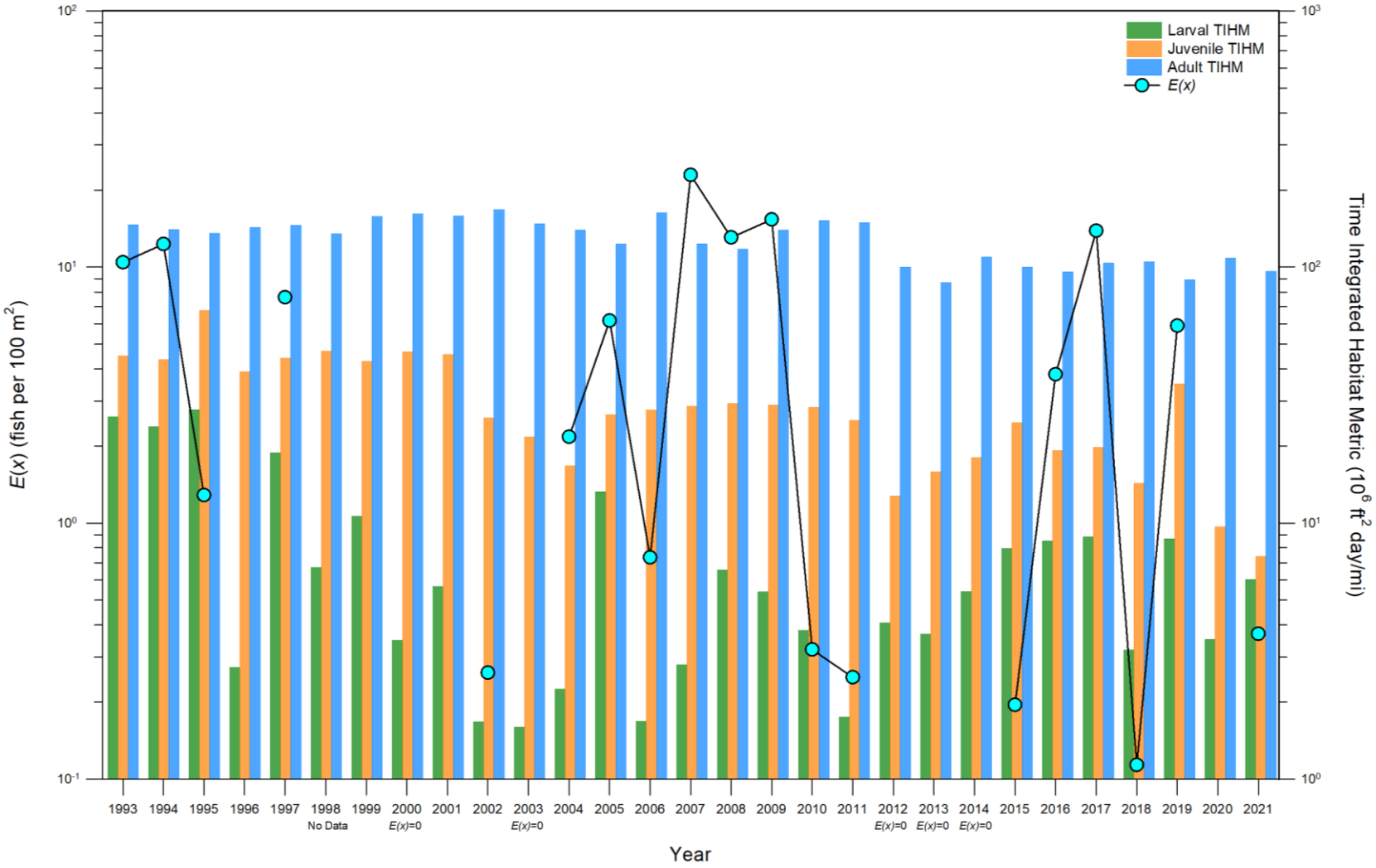


Figure 28. Annual Time Integrated Habitat Metrics (TIHMs), by life-stages of the Rio Grande Silvery Minnow, during the study period and annual estimated densities of the Rio Grande Silvery Minnow in October (*E(x)*; Angostura Reach). Sampling did not occur in 1998, *E*(*x*) could not be estimated for 1996, 1999, 2001 or 2020, and *E*(*x*) was zero in 2000, 2003, 2012, 2013, and 2014.Note the log scale of the y-axes.

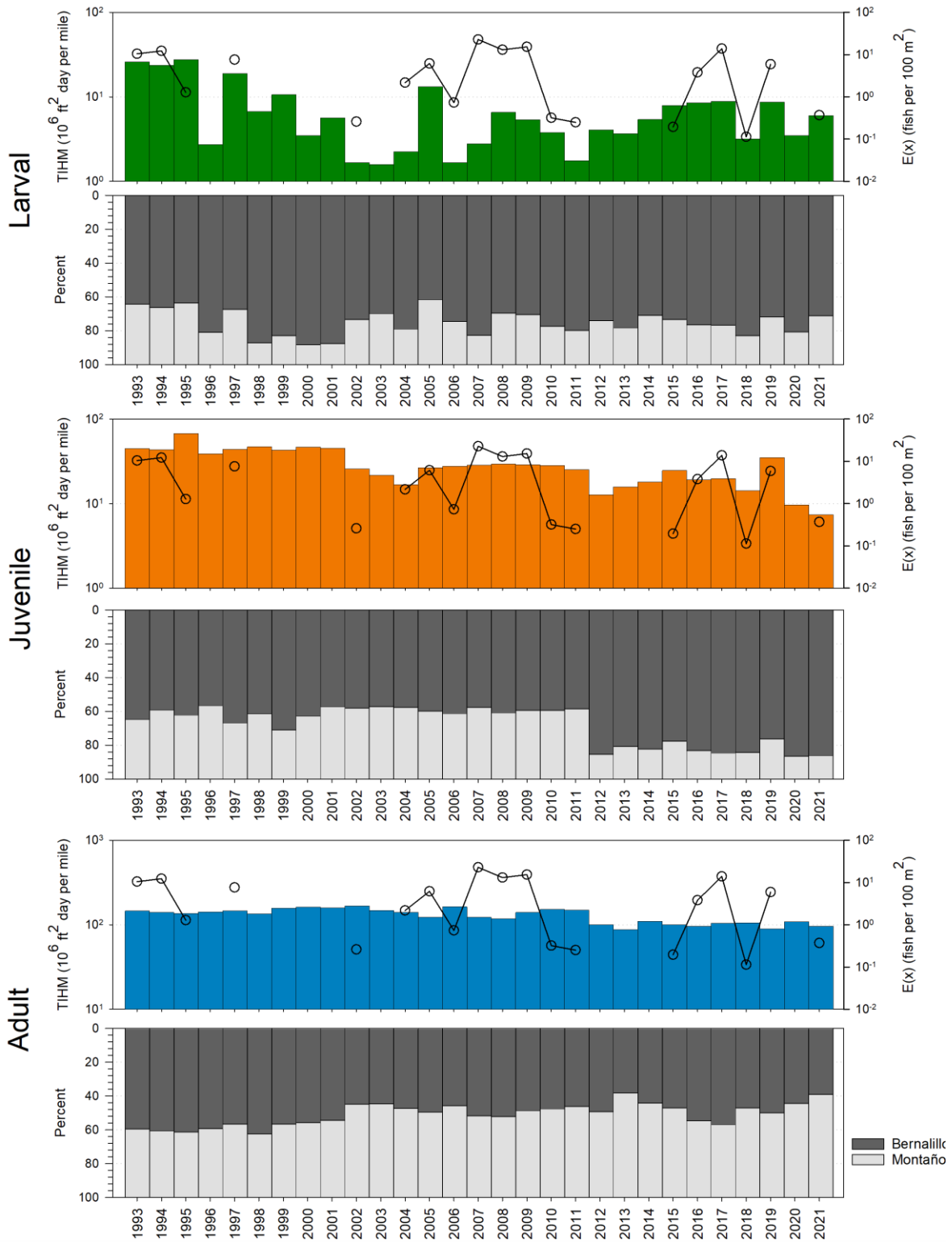


Figure 29. Annual Time Integrated Habitat Metrics (TIHMs; colored bars), by life-stages of the Rio Grande Silvery Minnow, during the study period (1993–2021), annual estimated densities of the Rio Grande Silvery Minnow in October (*E(x)*; black circles and lines), and percentage contribution by subreach (stacked bars). Note the log scale of the upper y-axes.

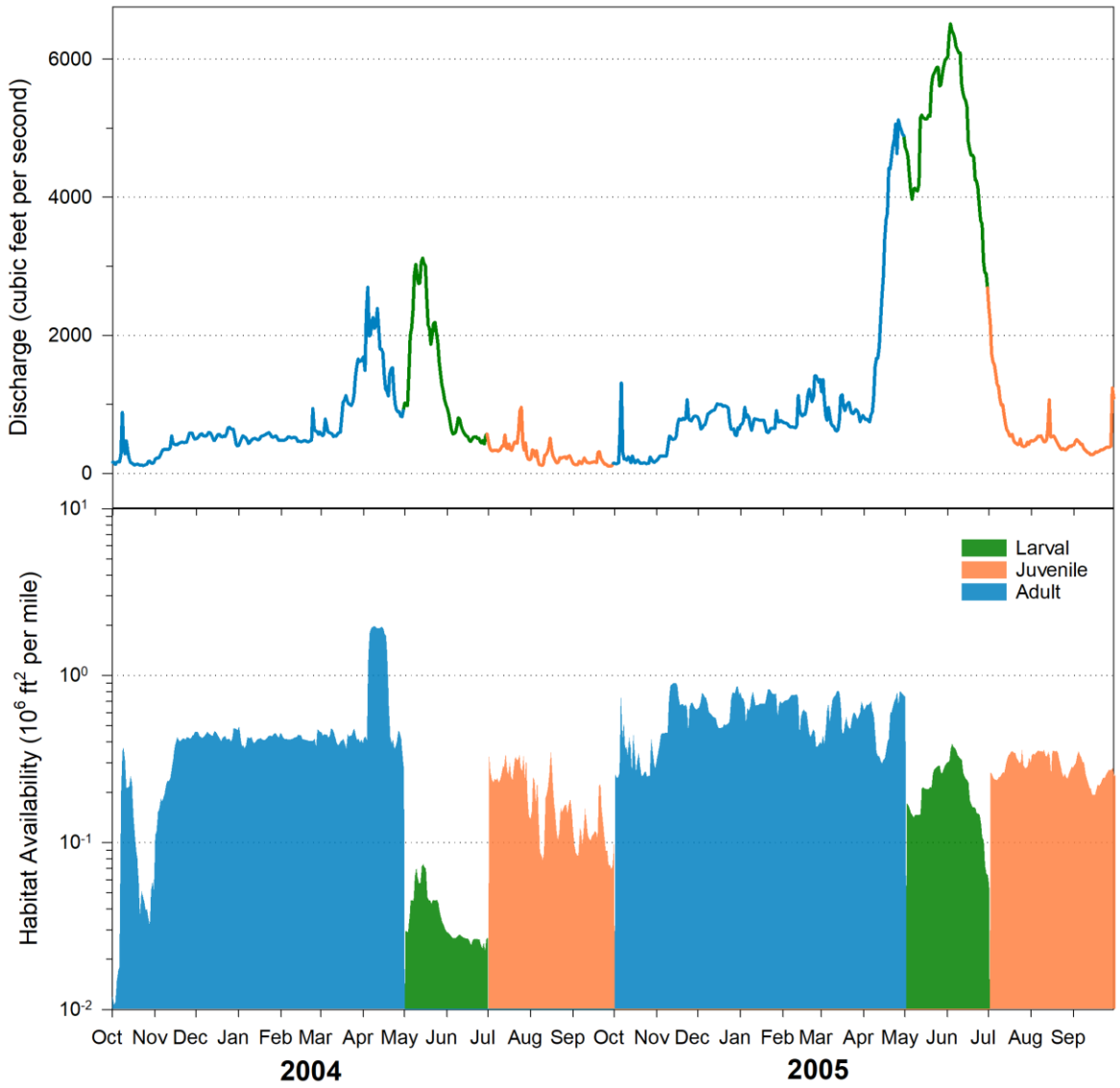


Figure 30. Hydrograph and habitat availability over time for water years 2004–2005. Colors correspond to life-stage periods; shaded areas (i.e., area under curve) represent Time Integrated Habitat Metrics (TIHMs). Note log-scale in bottom panel.

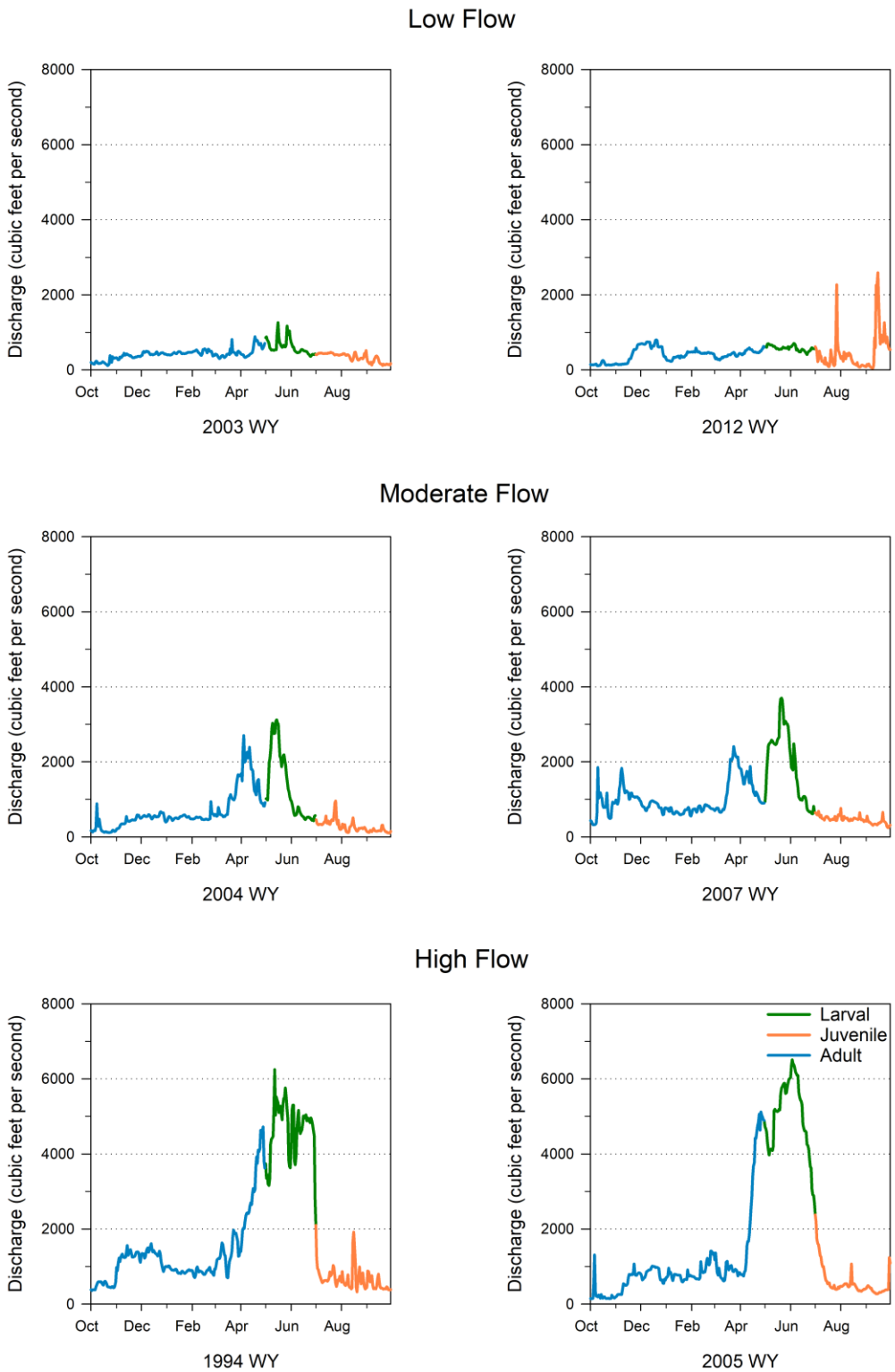


Figure 31. Selected annual hydrographs for assessment of geomorphic changes on Time Integrated Habitat Metrics (TIHMs). Hydrographs were selected to represent three flow scenarios: low, moderate, and high. Discharge data from USGS 08330000 Rio Grande at Albuquerque, NM.

Table 6. Time Integrated Habitat Metrics (TIHMs) for the Angostura Reach calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes on TIHMs over time (1962–2012).

			Time Integrated Habitat Metrics (10^6 ft^2 day/mi)		
Water Year	Flow Scenario	Survey Year	Larval	Juvenile	Adult
2003	low	1962	29.35	83.29	237.85
		1972	28.02	65.21	284.94
		1992	2.67	30.38	131.00
		2002	1.59	21.69	148.05
		2012	3.86	13.95	96.07
2012	low	1962	31.49	77.27	354.74
		1972	29.53	59.58	331.52
		1992	2.85	27.23	149.49
		2002	1.66	19.14	141.37
		2012	4.08	12.75	100.07
2004	moderate	1962	41.18	67.30	317.65
		1972	41.62	52.27	326.61
		1992	4.33	23.89	135.39
		2002	2.25	16.82	139.00
		2012	6.68	11.14	100.63
2007	moderate	1962	48.90	114.80	430.13
		1972	50.49	88.95	358.79
		1992	6.09	40.67	147.39
		2002	2.79	28.60	123.43
		2012	8.46	18.95	98.75
1994	high	1962	28.87	148.54	491.40
		1972	36.94	112.65	401.58
		1992	23.84	43.55	140.12
		2002	10.86	28.41	109.91
		2012	9.20	23.50	104.56
2005	high	1962	26.64	129.11	392.29
		1972	31.90	100.03	345.80
		1992	25.84	38.79	141.62
		2002	13.26	26.51	123.23
		2012	9.87	20.98	98.17

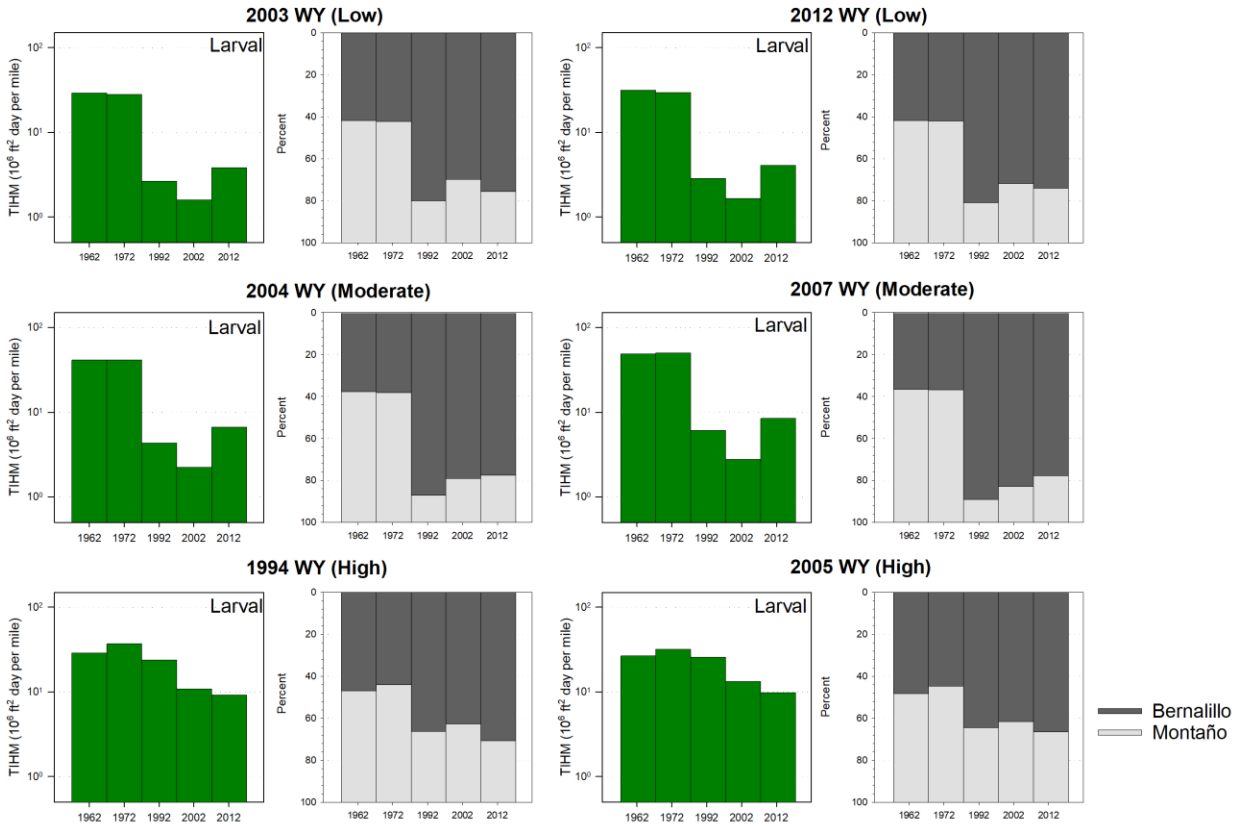


Figure 32. Time Integrated Habitat Metrics (TIHMs; colored bars) for the larval life-stage and percentage contribution by subreach (stacked bars) calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes over time.

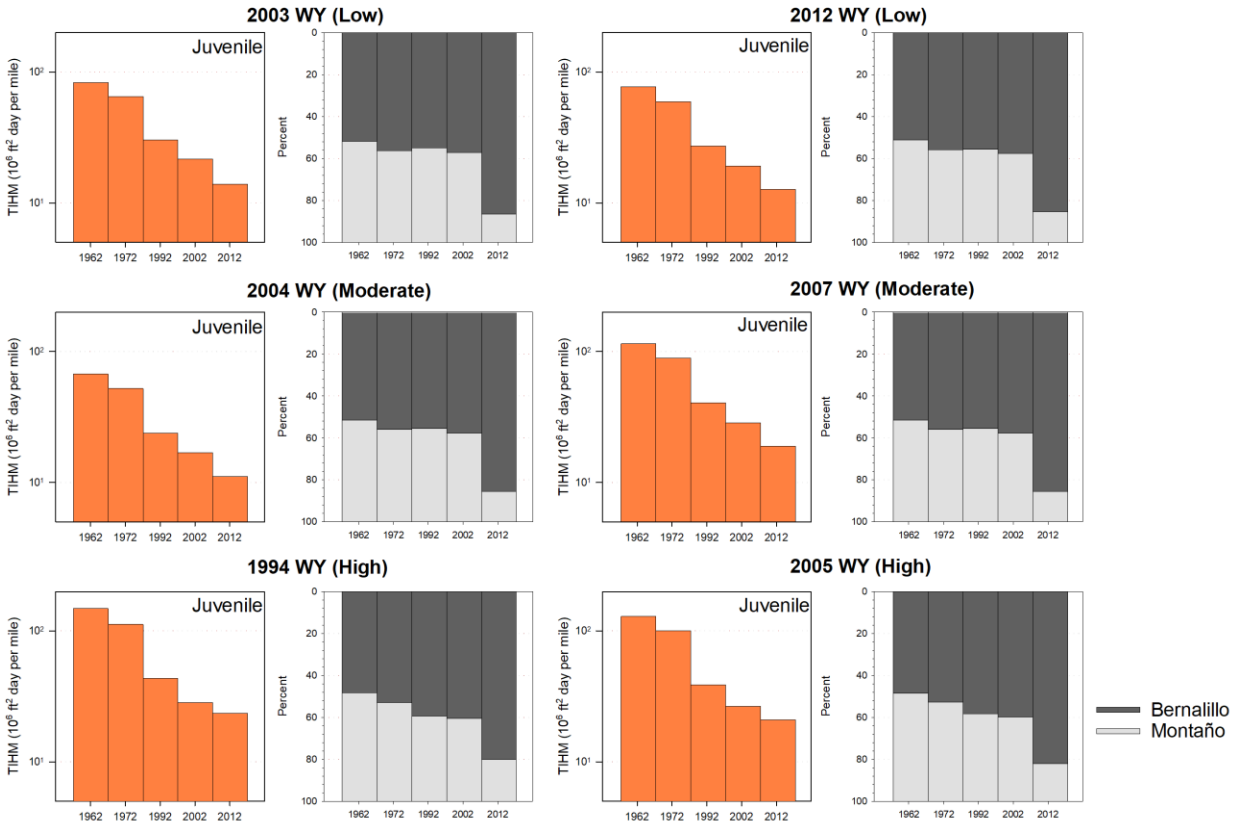


Figure 33. Time Integrated Habitat Metrics (TIHMs; colored bars) for the juvenile life-stage and percentage contribution by subreach (stacked bars) calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes over time.

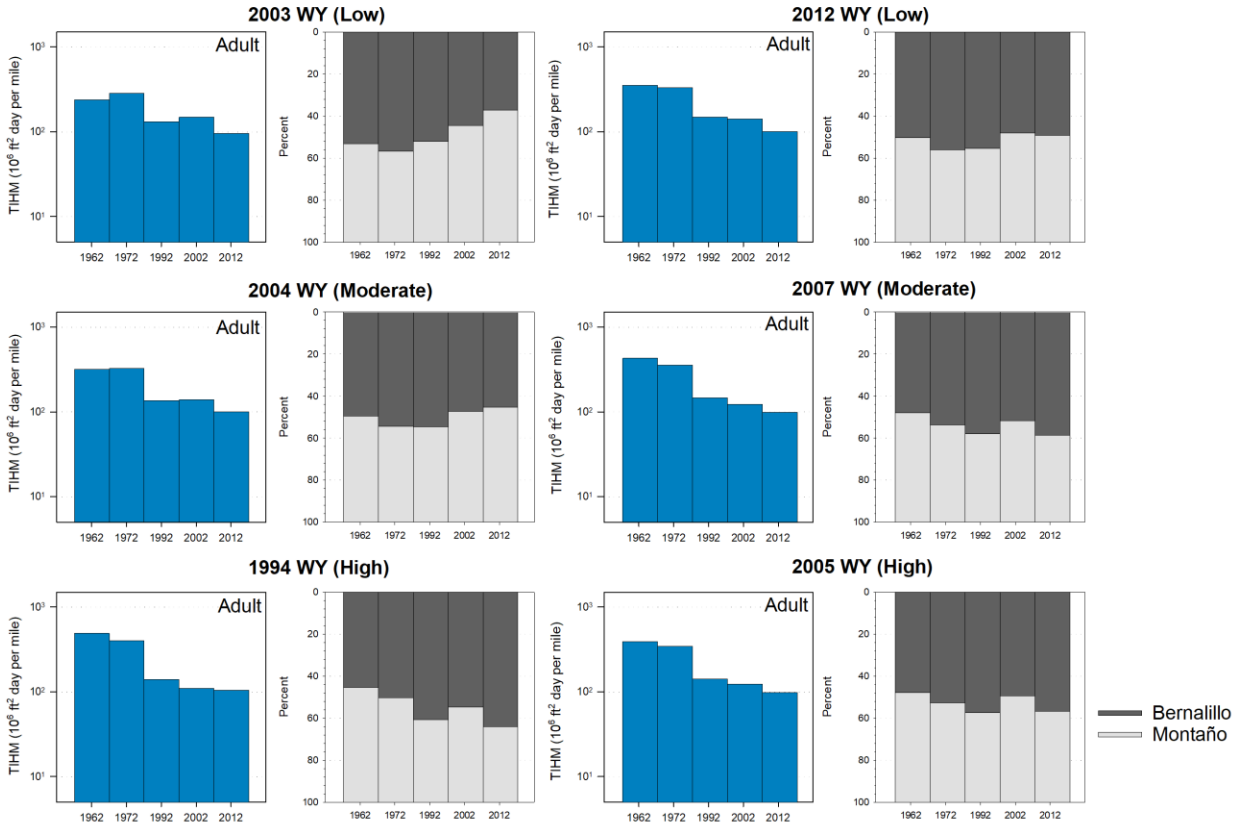


Figure 34. Time Integrated Habitat Metrics (TIHMs; colored bars) for the adult life-stage and percentage contribution by subreach (stacked bars) calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes over time.

Channel-Habitat Evolution Models

Stages of channel evolution were identified in the Angostura Reach based on designations for the Middle Rio Grande (Table 7, Figure 35; Massong et al., 2010). Stages of the planform evolution model were assigned for each subreach and survey year 1962–2012. This analysis indicated considerable temporal variation and minimal spatial variation in planform evolution stages in the study area. The Angostura Reach showed relatively homogenous trends in channel evolution as indicated by assessment at the subreach scale (e.g., Bernalillo and Montaño subreaches). All subreaches were characterized by the initial stages of the model in 1962 (Stage 1–3) and all evolved to stage M4–M5 (migrating stages) by 2012, with the exception of subreach B3 (stage 3).

The Bernalillo subreach (B1–B4) showed a more gradual shift to the migrating stages of the planform evolution model. This subreach was characterized as having progressed to stages 3–M4 in 2012 while the Montaño subreach was characterized as having reached M4 in 1992. By 2012, the Montaño subreach progressed to stage M5, which could be the final stage of planform evolution (i.e., arrested degradation) if conditions are stable. Trends observed 1962–2012 suggest the Angostura Reach will remain in or progress to stage M5 except perhaps for localized areas with potential for lateral mobility (e.g., no jetty-jacks, insufficient bed armoring), which could reflect dynamics associated with advanced migrating stages (M6–M8).

Stages of the planform evolution model were further evaluated using representative cross-sections, aerial imagery, and flow-habitat curves (Figures 36–37; Figures A-1–A-10). Cross-section data were used to illustrate the magnitude and rate of change in bed and bank elevations. Aerial imagery was used to show characteristic river planforms and temporal variation. Flow-habitat curves were used to assess interactions between discharge, channel morphology, and habitat availability, and how these relationships have changed through time and space. Additionally, Schied et al., (2022) illustrated cross-sections corresponding to the planform evolution stages described by Massong et al., (2010; Figure 34). These results further the existing planform evolution model by incorporating additional data and analyses, including relationships between flow and habitat availability, and through application of the model to relatively large spatial units (i.e., subreaches) over a long-term period of record. Modifications to the channel-habitat evolution model were used to create a more comprehensive view of channel evolution in the Middle Rio Grande and the subsequent impacts these processes have on the physical habitat conditions required by the Rio Grande Silvery Minnow.

Comparison of planforms, cross-sections, and flow-habitat curves across subreaches revealed spatially homogenous trends in channel evolution in the Angostura Reach. These trends were characterized by the development of a dominant single-threaded channel that progressively narrowed and incised. Channel incision corresponded to increased bankfull discharge, loss of floodplain connectivity, and decreased habitat availability. Jetty-jacks installations in the Angostura Reach ca. 1950–1980 stabilized and armored banks, establishing an engineered channel alignment from approximately the North Diversion Channel outfall to the Isleta Diversion Dam. Subsequent morphological changes have largely conformed to this channel alignment with colonization of vegetation stabilizing river forms (e.g., mid-channel bars and islands).

The Bernalillo subreach showed higher levels of habitat availability across flows for survey years 1992–2012 relative to the Montaño subreach. Morphological changes over time for each subreach are illustrated using representative cross-sections and planforms in this section (Figures 36–37; refer to Appendix A for channel-habitat evolution models for all subreaches [i.e., B1–B4, Mo1–Mo5]). Differences in habitat availability between subreaches were apparent in TIHMs as indicated by unequal percent contributions (Figures 30–32). However, differences in habitat availability between subreaches seemed to have minimal impact on habitat availability at the reach-scale (Figure 24).

Table 7. Classification of Angostura subreaches based on the planform evolution model for the Middle Rio Grande (Massong et al., 2010). Migrating stages (M) are indicative of excessive sediment transport capacity.

Subreach	1962	1972	1992	2002	2012
B1	1	1	2	M4	M4
B2	1	1	2	3	M4
B3	1	1	2	3	3
B4	1	1	2	3	M4
Mo1	2	3	M4	M4	M5
Mo2	2	3	M4	M5	M5
Mo3	3	3	M4	M5	M5
Mo4	3	3	M4	M5	M5
Mo5	2	3	M4	M5	M5

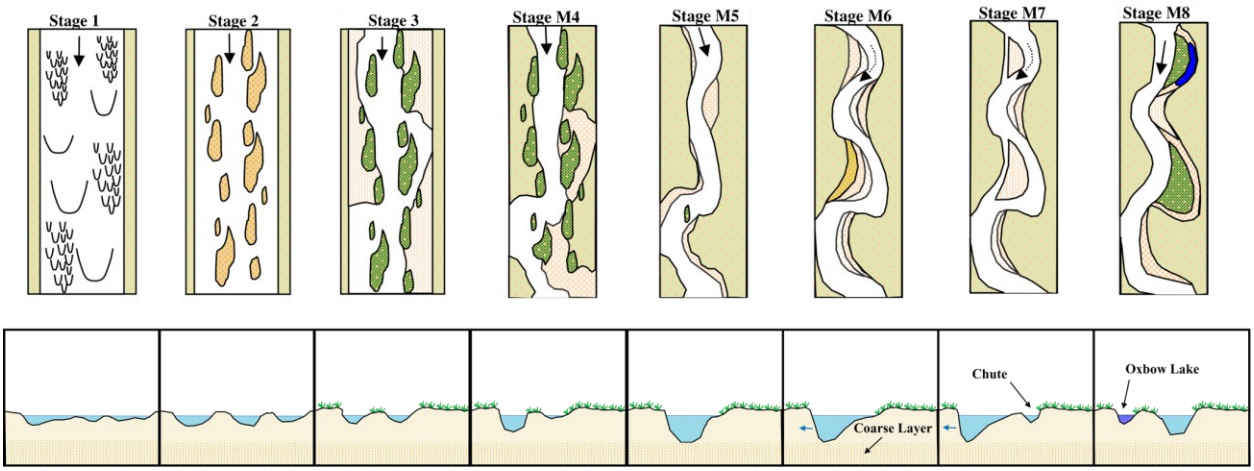


Figure 35. Planform evolution model for the Middle Rio Grande including characteristic cross-sections (Schied et al., 2023; modified from Massong et al., 2010). Only migrating (M) stages are illustrated; only migrating (M) stages were observed in the Angostura Reach (i.e., excessive sediment transport capacity).

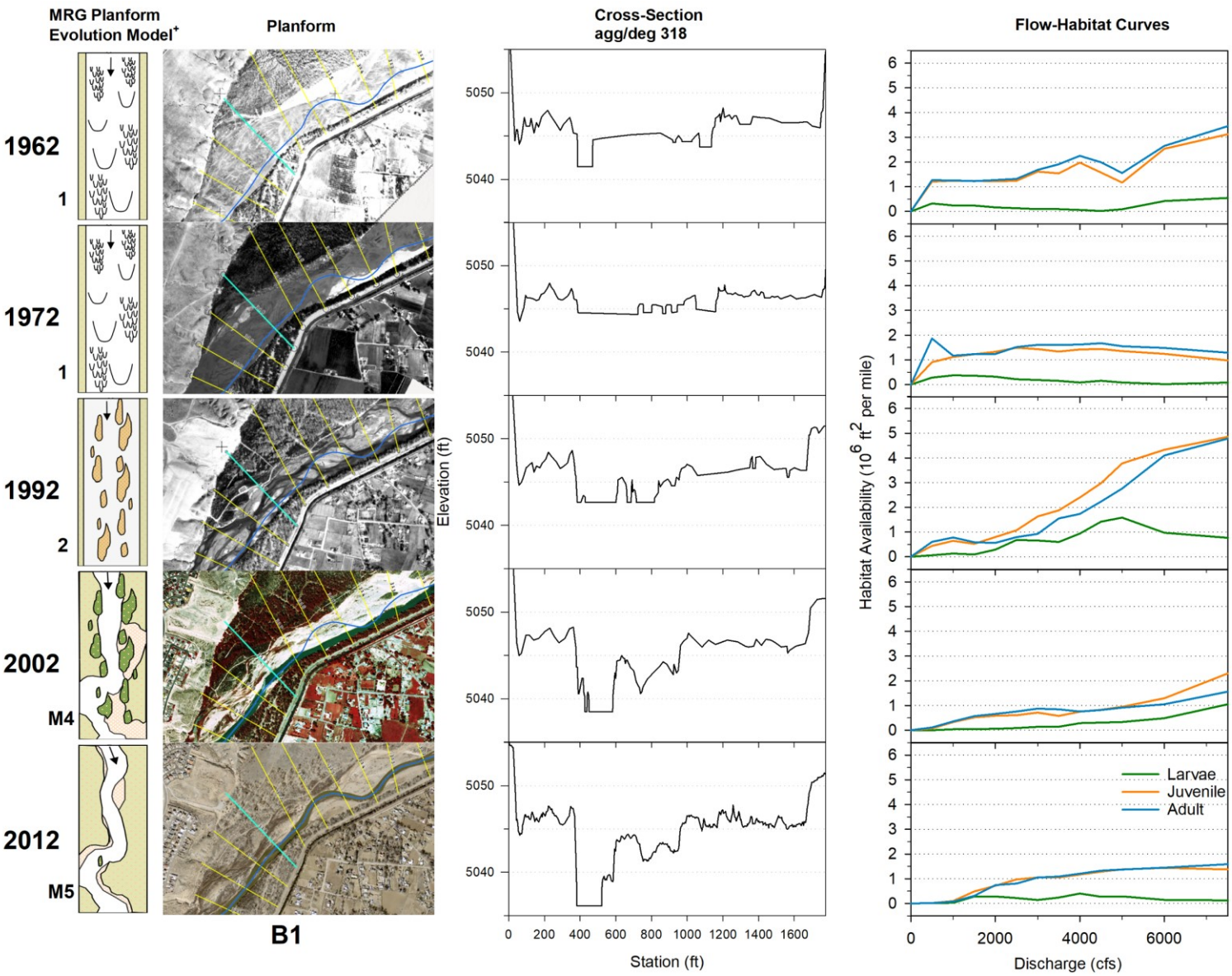


Figure 36. Channel-habitat evolution model for subreach B1. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

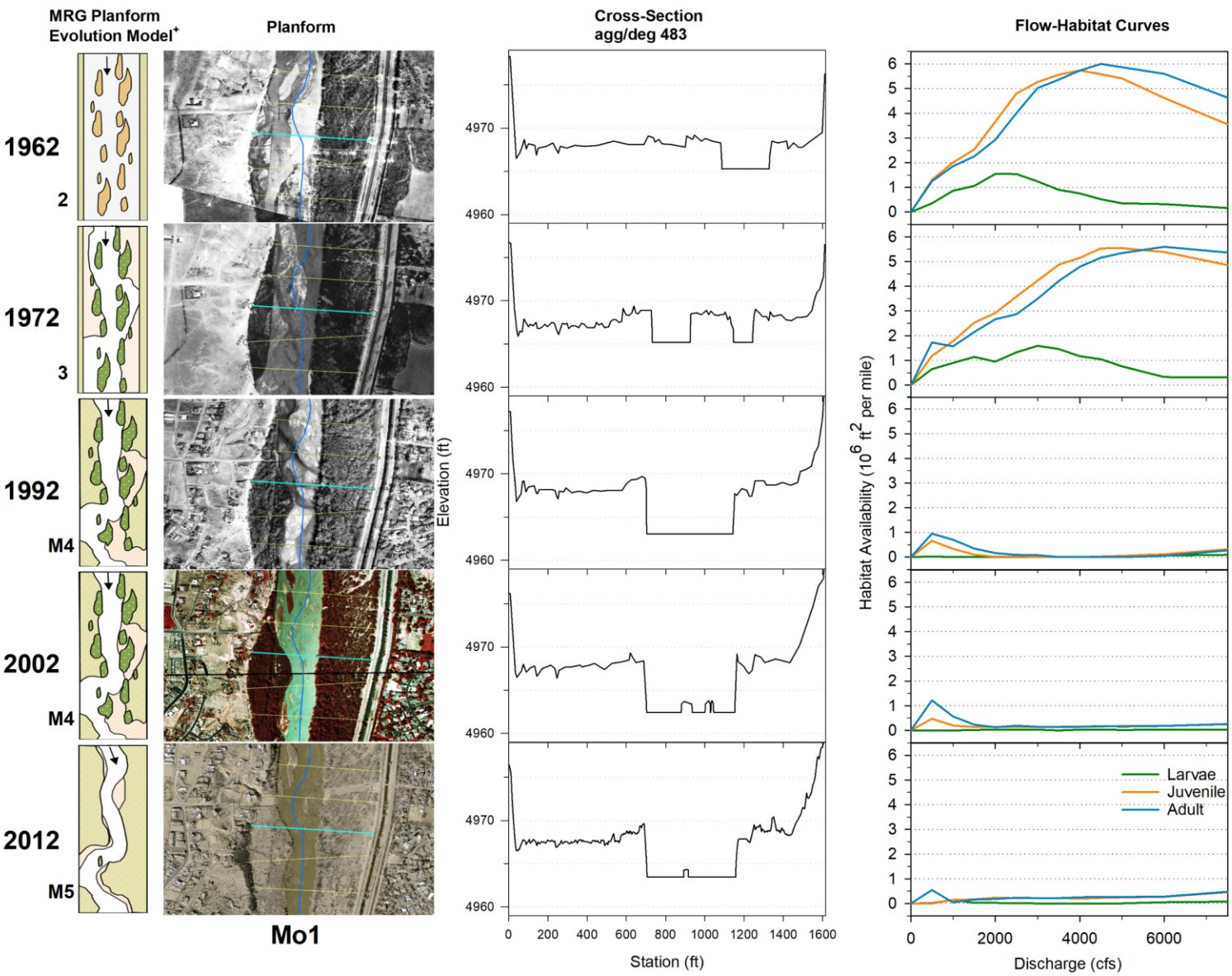


Figure 37. Channel-habitat evolution model for subreach Mo1. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

Rio Grande Silvery Minnow Population Analyses (1993–2021)

Reach-Scale Population Data

Rio Grande Silvery Minnow population parameters (i.e., estimated occurrence probability, estimated lognormal density, and estimated density) were computed for the Angostura Reach. Rio Grande Silvery Minnow densities ($E(x)$; estimated using October sampling-site data [1993–2021]) were generated from the year model (δ[Year] μ[Year]). Estimated density was notably higher ($P < 0.05$) in 2017, as compared with 2018, but then increased over tenfold from 2018 to 2019 (Figure 38). However, recent monitoring efforts revealed a substantial decrease (–98.8%) in the density of Rio Grande Silvery Minnow from 2019 ($E(x) = 5.90$) to 2020 ($E(x) = 0.07$), and its density remained low in 2021 ($E(x) = 0.37$). Naïve density estimates (i.e., unmodeled), calculated using the method of moments, were very similar to model-estimated densities ($E(x)$). Combining a plot of $E(x)$ and mean daily discharge (1993–2021) revealed a long-term recurrent pattern of increased densities during years with high spring runoff and decreased densities during years with low spring runoff (Figure 39). Estimates of $E(x)$ were generally highest when habitat and flow values were elevated (Figure 40: A–I). For brevity, trends in reach-scale estimated density are discussed herein; reach-scale occurrence probability (δ) and lognormal density (μ) for the study period are presented in the following section.

Long-Term Ecological Relationships

Relationships between Rio Grande Silvery Minnow population parameters and environmental covariates (i.e., habitat and flow metrics; Tables 5 and 8) were evaluated using robust statistical methods. Habitat and flow metrics were evaluated independently (Table 9, 10) and together (Table 11). The occurrence probability (δ) and the lognormal density (μ), estimated from the year model (δ[Year] μ[Year]), were closely associated with environmental covariates over time (1993–2021). Estimates of δ increased with elevated habitat and flow values (Figures 41–42). Similar and consistent results were obtained for relationships between μ and habitat/flow metrics (Figures 43–44).

Generalized linear models of Rio Grande Silvery Minnow mixture-model estimates revealed that variation in both δ and μ was strongly predicted by changes in habitat metrics (TIHMs) across years (1993–2021 [TIHMs analyzed independently of flow metrics]; Table 9). The top ecological model (δ[MayJunHab+R] μ[MayJunHab+R]) received 56.9% of the AIC_c weight (w_i) out of the 64 models considered. The top δ covariate (MayJunHab) accounted for 17.2% of the deviance ($P < 0.05$) explained by the δ(Year) model over the null model, δ(.). However, we found no significant effects for JulSepHab (9.0%) or OctAprHab (< 0.1%). Further, the top μ covariate (MayJunHab) accounted for only 4.7% of the deviance ($P > 0.05$) explained by the μ(Year) model over the null model, μ(.). Similarly, we found no significant effects for JulSepHab (2.4%) or OctAprHab (1.1%). The top three habitat models, which accounted for most of the cumulative w_i (ca. 85%), were based on metrics representing elevated larval (May–June) and juvenile (July–September) habitat availability. In summary, higher larval TIHM values (i.e., more larval fish habitat during spring [May–June]) best predicted the increased occurrence and density of Rio Grande Silvery Minnow (based on October sampling) over time.

Similarly, generalized linear models of Rio Grande Silvery Minnow mixture-model estimates revealed that variation in both δ and μ was also strongly predicted by changes in flow metrics across years (1993–2021 [flow metrics analyzed independent of TIHMs]; Table 10). The top ecological model (δ[MayJun28dHigh+R] μ[MayJun28dHigh+R]) received 55.0% of the AIC_c weight (w_i) out of the 196 models considered. The top δ covariate (MayJun28dHigh) accounted for 49.5% of the deviance ($P < 0.001$) explained by the δ(Year) model over the null model, δ(.). We also found significant effects for MayJunMean (46.5%; $P < 0.001$), OctAprMean (25.2%; $P < 0.01$), and JulSepMean (15.5%; $P < 0.05$), but not for JulSep7dLow (7.4%) or OctApr7dLow (2.6%). Further, the top μ covariate (MayJun28dHigh) accounted for 22.0% of the deviance ($P < 0.01$) explained by the μ(Year) model over the null model, μ(.). We also found a significant effect for OctAprMean (16.4%; $P < 0.05$), but not for MayJunMean (13.1%), JulSep7dLow (6.4%), JulSepMean (2.1%), or OctApr7dLow (1.0%). The top five flow models, which accounted for nearly all the cumulative w_i (ca. 98%), were based on metrics representing elevated flows during spring. In summary, higher spring flows (ca. one month in duration) best predicted the increased occurrence and density of Rio Grande Silvery Minnow over time.

Finally, generalized linear models of Rio Grande Silvery Minnow mixture-model estimates revealed that variation in both δ and μ was better predicted by changes in flow metrics, as compared to TIHMs, across years (1993–2021 [TIHMs and flow metrics analyzed together]; Table 11). The top ecological model (δ[MayJun28dHigh+*R*] μ[MayJun28dHigh+*R*]) received 54.9% of the AIC_c weight (w_i) out of the 400 models considered. The top five models, which accounted for nearly all the cumulative w_i (ca. 98%), were based on metrics representing elevated flows during spring. As compared to all other habitat and flow metrics, we found that higher spring flows (ca. one month in duration) best predicted the increased occurrence and density of Rio Grande Silvery Minnow over time.

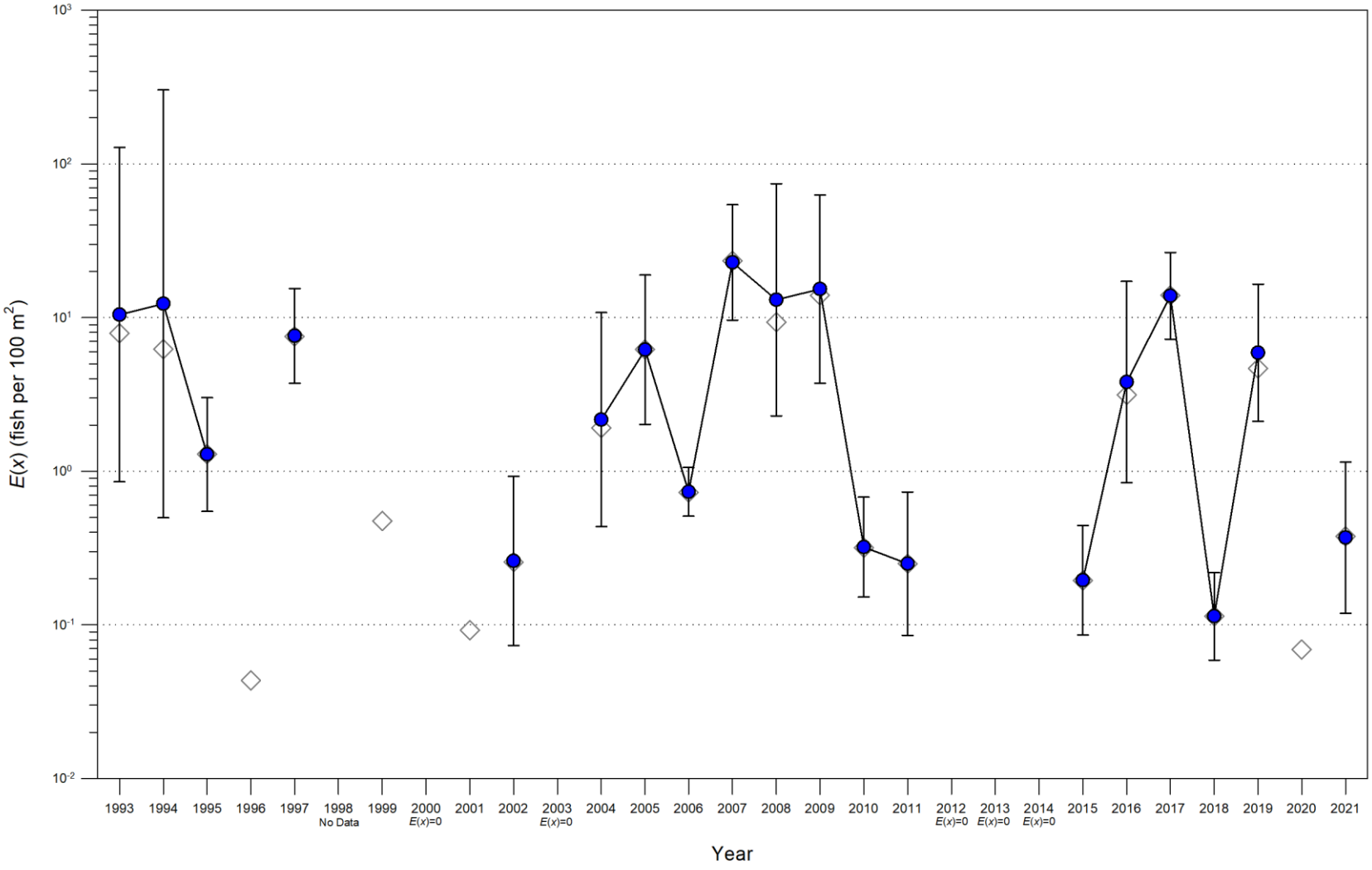


Figure 38. Densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data) from the Angostura Reach across years. Sampling did not occur in 1998, $E(x)$ could not be estimated for 1996, 1999, 2001, or 2020, and E(x) was zero for 2000, 2003, 2012, 2013, and 2014. Modeled estimates (circles), 95% confidence intervals (bars), and method-of-moments estimates (diamonds) are illustrated.

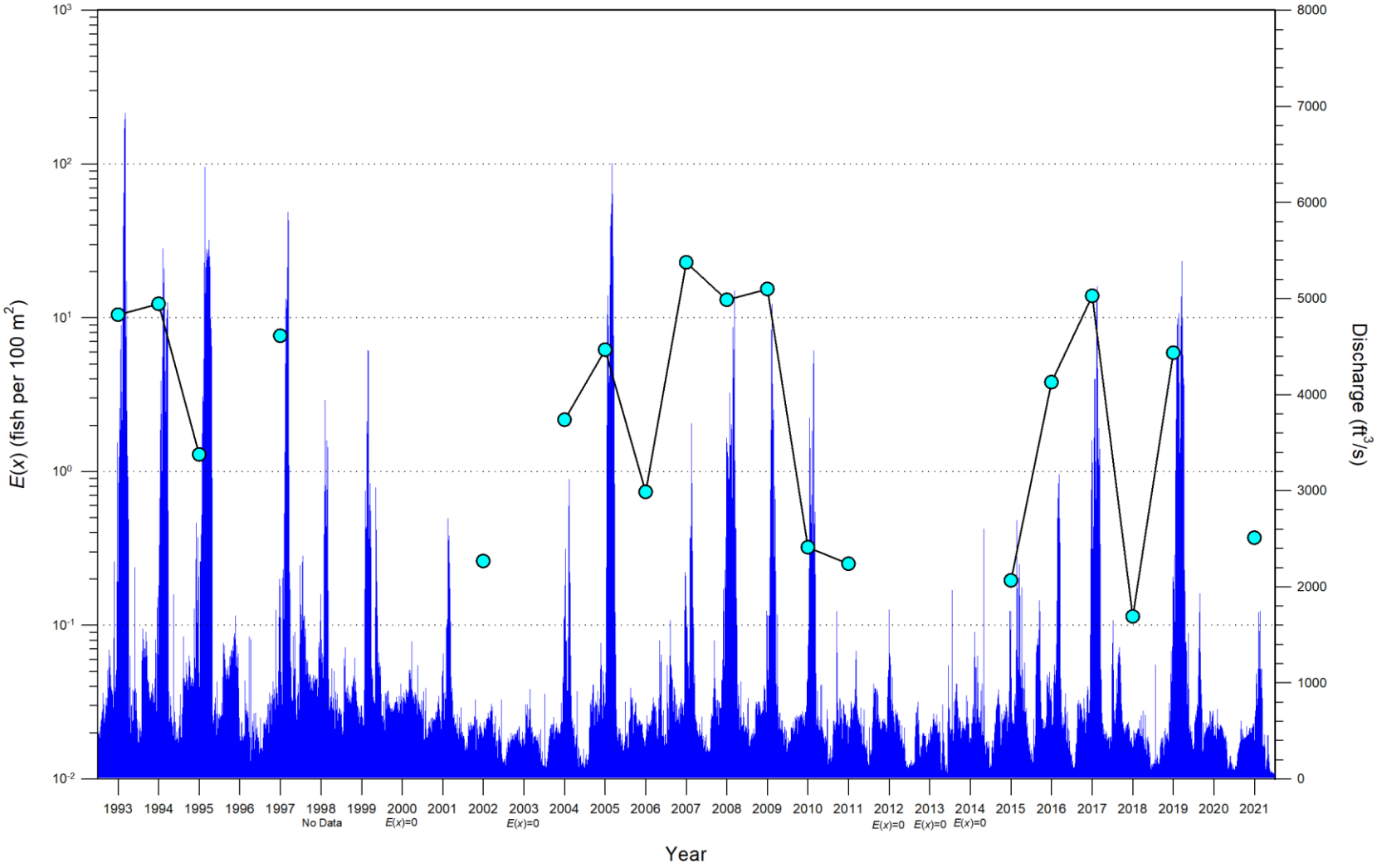


Figure 39. Densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data from the Angostura Reach) and mean daily discharge data from the Angostura Reach across years. Sampling did not occur in 1998, $E(x)$ could not be estimated for 1996, 1999, 2001 or 2020, and $E(x)$ was zero in 2000, 2003, 2012, 2013, and 2014.

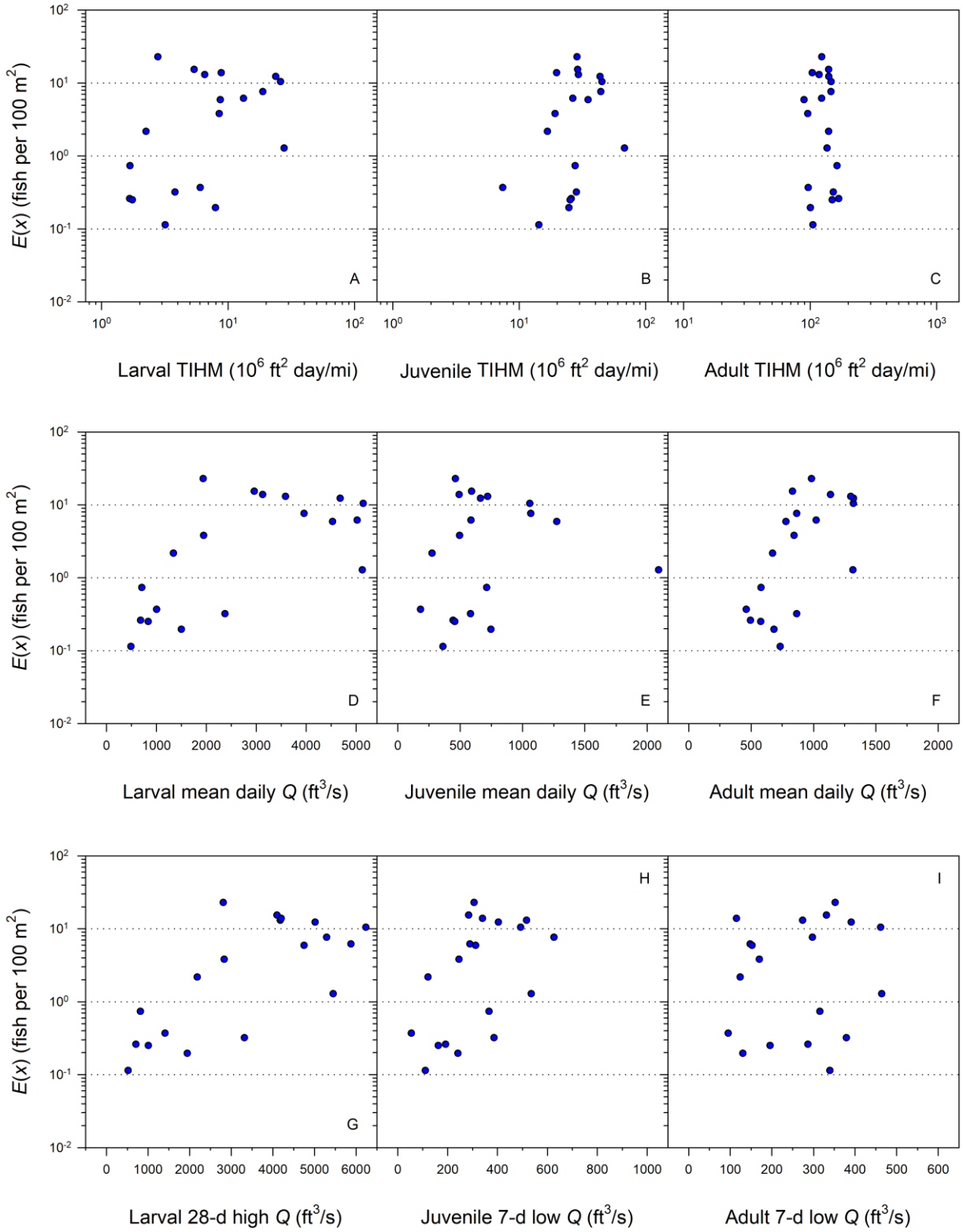


Figure 40. Bivariate plots of densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data from the Angostura Reach), TIHMs (panels A–C), and flow metrics (panels D–I).

Table 8. Flow metrics for the Angostura Reach 1993–2021. Discharge data were obtained from the Rio Grande at Albuquerque, NM (USGS 08330000).

	Angostura Reach Flow Metrics (cubic feet per second)					
Water Year	Larval[1] mean daily	Juvenile[2] mean daily	Adult[3] mean daily	Larval 28-d high	Juvenile 7-d low	Adult 7-d low
1993	5143	1060	1321	6234	493	462
1994	4681	663	1321	5015	404	391
1995	5126	2092	1317	5450	535	464
1996	645	493	1052	732	289	398
1997	3960	1067	866	5291	627	298
1998	2286	764	1300	3248	560	935
1999	3133	1377	837	3760	770	477
2000	987	810	773	1095	700	576
2001	1826	585	677	2468	428	390
2002	683	443	496	706	192	287
2003	615	332	413	725	142	151
2004	1341	274	672	2180	121	124
2005	5021	586	1021	5878	289	148
2006	707	714	580	817	366	316
2007	1937	461	984	2806	305	352
2008	3587	721	1299	4178	517	274
2009	2960	592	832	4100	284	331
2010	2374	582	866	3317	386	379
2011	837	456	578	1006	162	196
2012	653	315	720	674	113	251
2013	578	458	397	608	91	128
2014	889	482	659	1074	120	406
2015	1500	747	683	1944	241	130
2016	1947	496	844	2831	245	170
2017	3130	492	1137	4199	340	115
2018	490	361	734	521	110	339
2019	4528	1275	780	4749	312	152
2020	545	233	669	562	101	289
2021	1004	182	462	1409	54	95

[1] = Larval life-stage corresponds to May–June
[2] = Juvenile life-stage corresponds to July–September
[3] = Adult life-stage corresponds to October–April

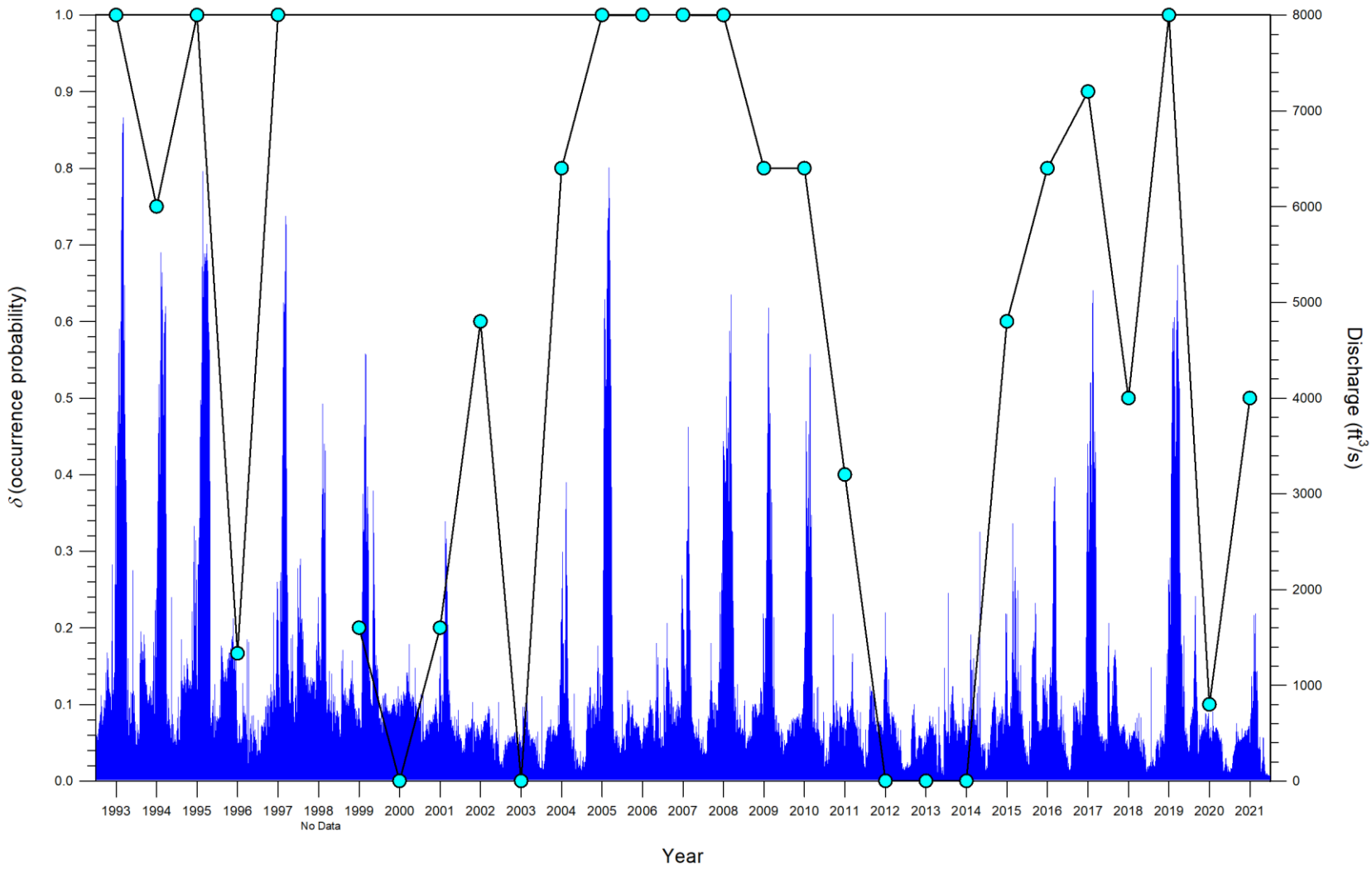


Figure 41. Occurrence probabilities of the Rio Grande Silvery Minnow (δ; estimated using October sampling-site data from the Angostura Reach) and mean daily discharge data from the Angostura Reach across years. Sampling did not occur in 1998.

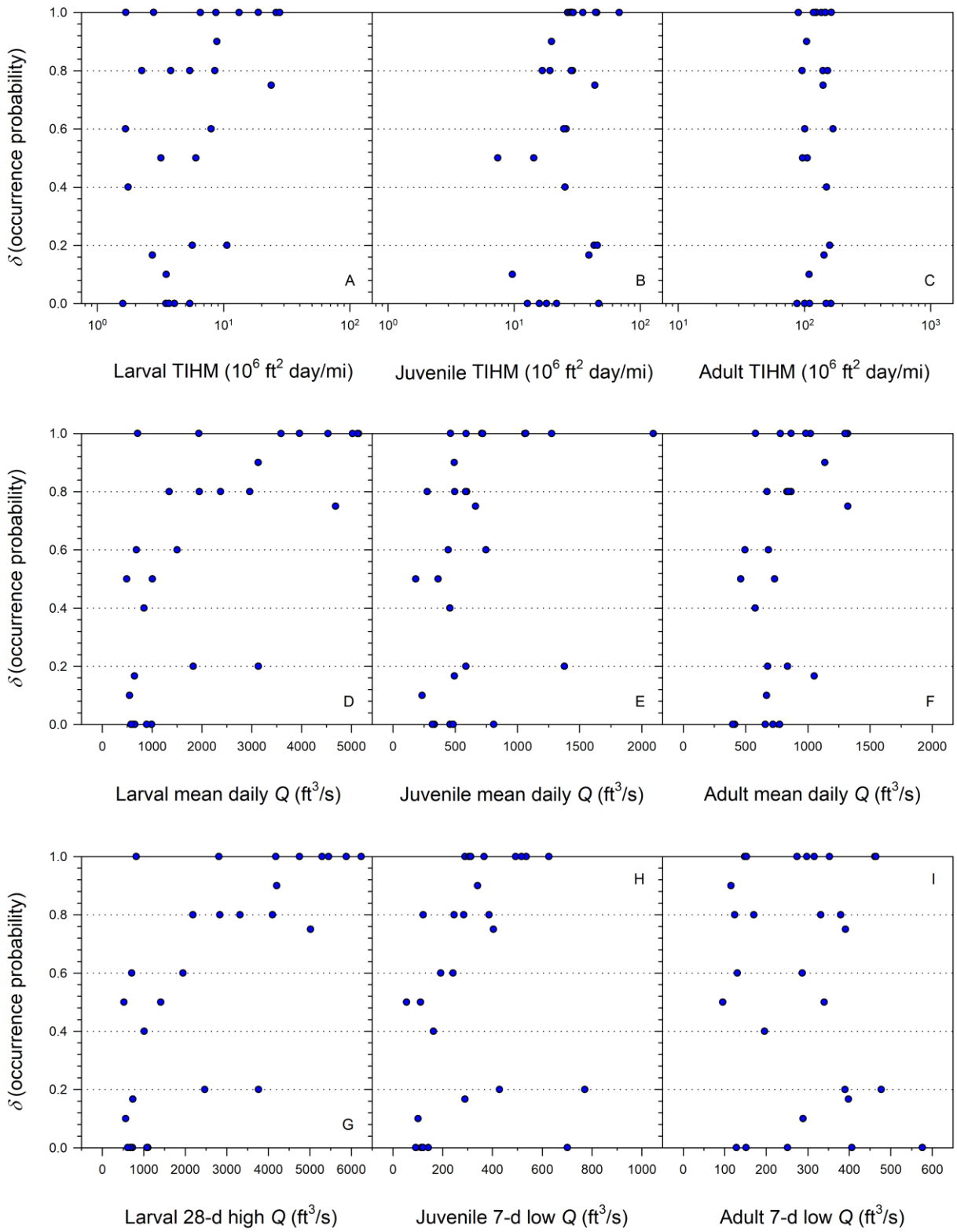


Figure 42. Bivariate plots of occurrence probabilities of the Rio Grande Silvery Minnow (δ; estimated using October sampling-site data from the Angostura Reach), TIHMs (panels A–C), and flow metrics (panels D–I).

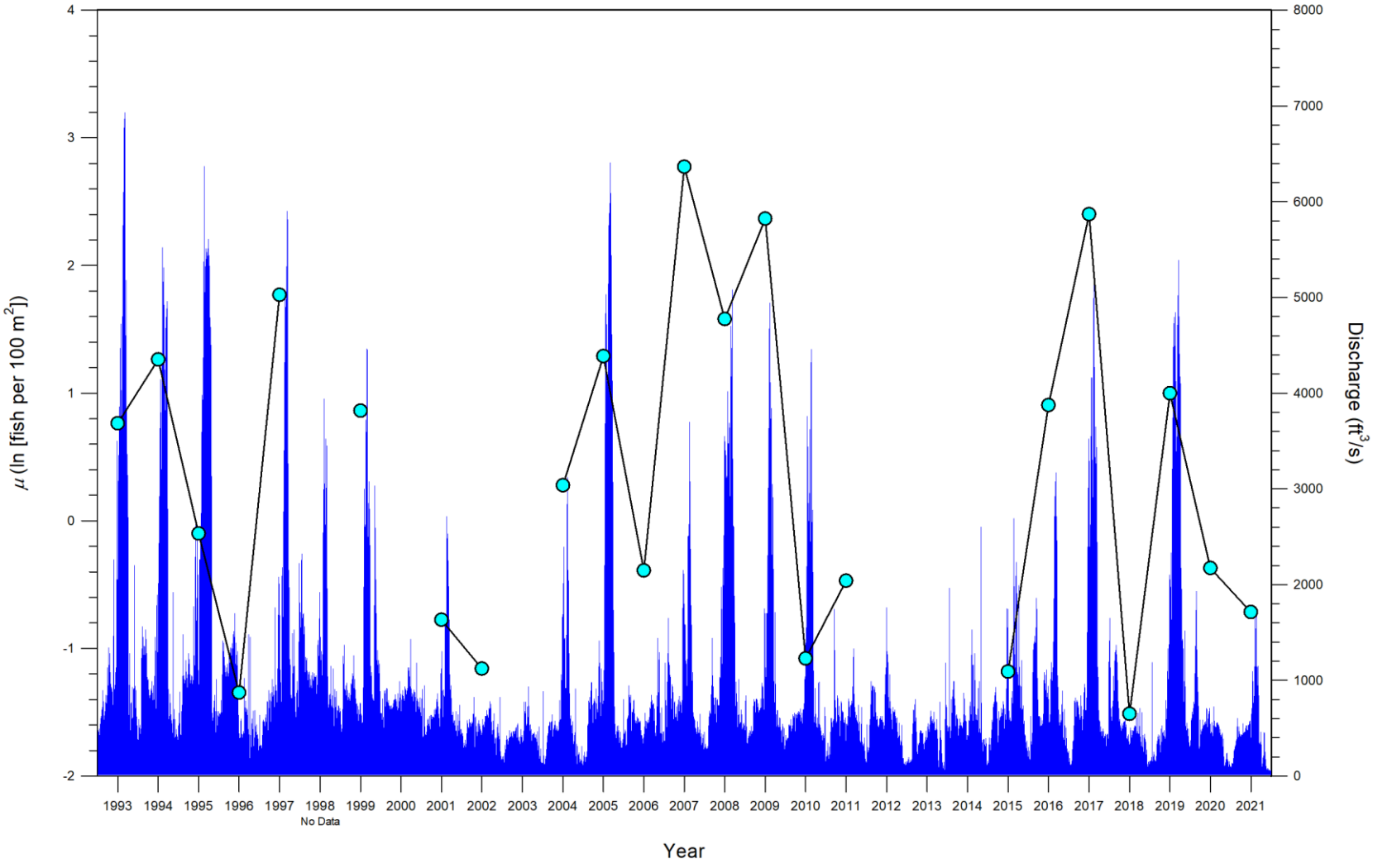


Figure 43. Lognormal densities of the Rio Grande Silvery Minnow (μ; estimated using October sampling-site data from the Angostura Reach) and mean daily discharge data from the Angostura Reach across years. Sampling did not occur in 1998, and μ could not be estimated for 2000, 2003, 2012, 2013, or 2014 (i.e., all zero values).

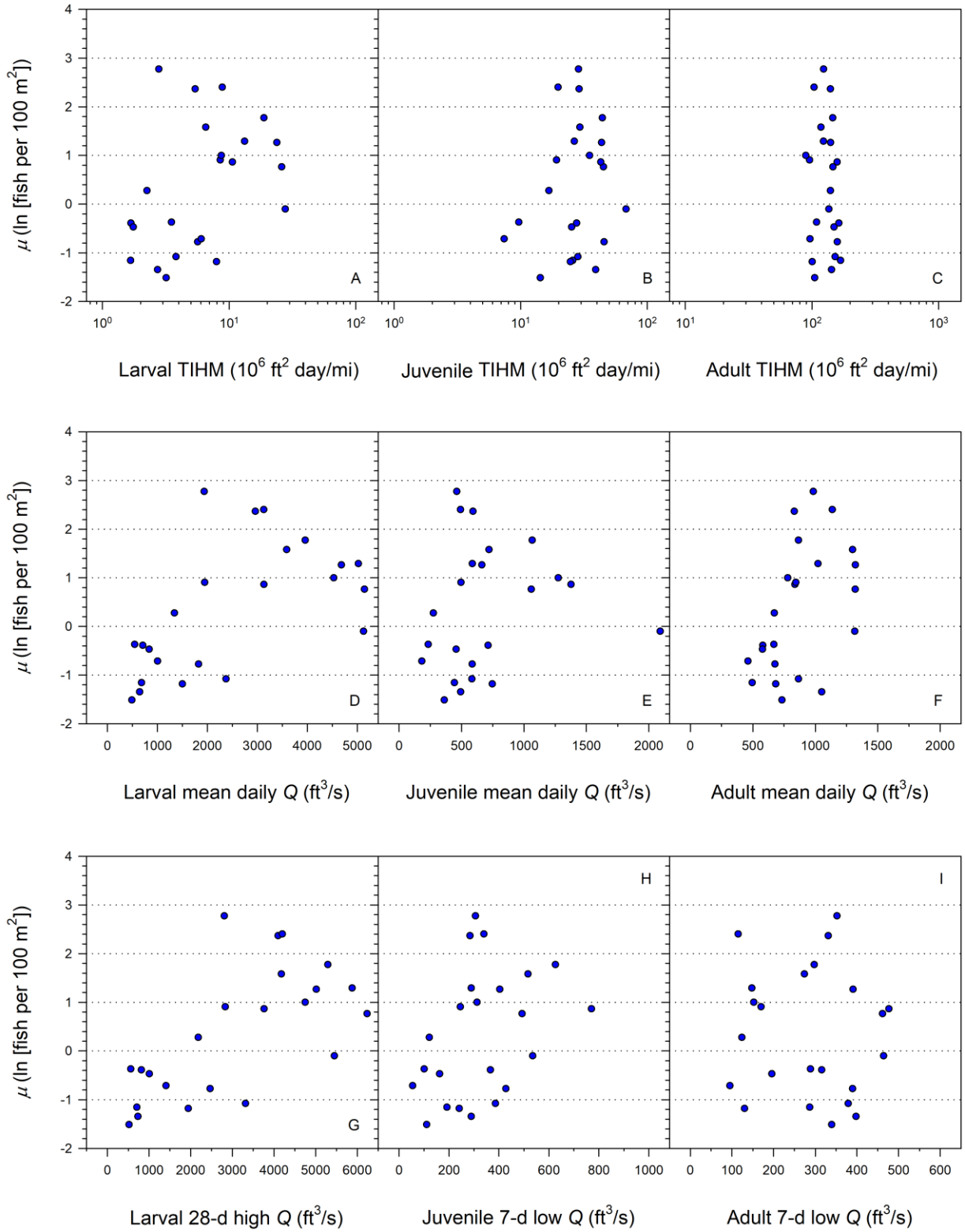


Figure 44. Bivariate plots of lognormal densities of the Rio Grande Silvery Minnow (μ; estimated using October sampling-site data from the Angostura Reach), TIHMs (panels A–C), and flow metrics (panels D–I).

Table 9. Generalized linear models of mixture-model estimates for the Rio Grande Silvery Minnow, using October sampling-site data from the Angostura Reach (1993–2021). Only Time Integrated Habitat Metrics (TIHMs: MayJunHab, JulSepHab, and OctAprHab) were included as modeling covariates.

Model[1]	**logLike**[2]	**K**[3]	**AIC$_c$**[4]	***w$_i$***[4]
δ(MayJunHab+*R*) μ(MayJunHab+*R*)	308.81	9	327.97	0.5693
δ(*R*) μ(MayJunHab+*R*)	313.85	8	330.78	0.1403
δ(JulSepHab+*R*) μ(MayJunHab+*R*)	311.67	9	330.83	0.1364
δ(OctAprHab+*R*) μ(MayJunHab+*R*)	313.79	9	332.96	0.0472
δ(*R*) μ(*R*)	321.18	6	333.71	0.0323
δ(JulSepHab+*R*) μ(*R*)	319.88	7	334.59	0.0208
δ(JulSepHab+*R*) μ(JulSepHab+*R*)	315.73	9	334.89	0.0179
δ(*R*) μ(JulSepHab+*R*)	318.72	8	335.64	0.0123
δ(MayJunHab+*R*) μ(JulSepHab+*R*)	317.15	9	336.31	0.0088
δ(OctAprHab+*R*) μ(JulSepHab+*R*)	318.48	9	337.64	0.0045

[1] = Models included all δ and μ combinations of null effects (.), random effects (*R*), and Time Integrated Habitat Metrics (with and without *R*).
[2] = Likelihood (–2[log-likelihood]) was estimated for each model.
[3] = Higher numbers of parameters indicate increased model complexity.
[4] = Top ten models were ranked (best to worst) by Akaike's information criterion (AIC$_c$) and include the AIC$_c$ weight (w_i); lower AIC$_c$ and higher w_i indicate a better model.

Table 10. Generalized linear models of mixture-model estimates for the Rio Grande Silvery Minnow, using October sampling-site data from the Angostura Reach (1993–2021). Only flow metrics (MayJunMean, JulSepMean, OctAprMean, MayJun28dHigh, JulSep7dLow, and OctApr7dLow) were included as modeling covariates.

Model[1]	**logLike**[2]	**K**[3]	**AIC$_c$**[4]	**w_i**[4]
δ(MayJun28dHigh+*R*) μ(MayJun28dHigh+*R*)	286.45	9	305.61	0.5501
δ(MayJunMean+*R*) μ(MayJun28dHigh+*R*)	287.48	9	306.65	0.3284
δ(MayJun28dHigh+*R*) μ(MayJunMean+*R*)	290.86	9	310.03	0.0606
δ(MayJunMean+*R*) μ(MayJunMean+*R*)	291.94	9	311.10	0.0355
δ(MayJun28dHigh) μ(MayJun28dHigh+*R*)	299.29	7	314.00	0.0083
δ(MayJun28dHigh+*R*) μ(OctAprMean+*R*)	295.52	9	314.69	0.0059
δ(MayJunMean+*R*) μ(OctAprMean+*R*)	297.16	9	316.33	0.0026
δ(OctAprMean+*R*) μ(MayJun28dHigh+*R*)	297.46	9	316.62	0.0022
δ(JulSepMean+*R*) μ(MayJun28dHigh+*R*)	297.78	9	316.94	0.0019
δ(MayJunMean) μ(MayJun28dHigh+*R*)	302.96	7	317.67	0.0013

[1] = Models included all δ and μ combinations of null effects (.), random effects (*R*), and flow metrics (with and without *R*).
[2] = Likelihood (–2[log-likelihood]) was estimated for each model.
[3] = Higher numbers of parameters indicate increased model complexity.
[4] = Top ten models were ranked (best to worst) by Akaike's information criterion (AIC$_c$) and include the AIC$_c$ weight (w_i); lower AIC$_c$ and higher w_i indicate a better model.

Table 11. Generalized linear models of mixture-model estimates for the Rio Grande Silvery Minnow, using October sampling-site data from the Angostura Reach (1993–2021). Time Integrated Habitat Metrics (TIHMs: MayJunHab, JulSepHab, and OctAprHab) and flow metrics (MayJunMean, JulSepMean, OctAprMean, MayJun28dHigh, JulSep7dLow, and OctApr7dLow) were both included as modeling covariates.

Model[1]	**logLike**[2]	**K**[3]	**AIC$_c$**[4]	**w_i**[4]
δ(MayJun28dHigh+*R*) μ(MayJun28dHigh+*R*)	286.45	9	305.61	0.5486
δ(MayJunMean+*R*) μ(MayJun28dHigh+*R*)	287.48	9	306.65	0.3275
δ(MayJun28dHigh+*R*) μ(MayJunMean+*R*)	290.86	9	310.03	0.0604
δ(MayJunMean+*R*) μ(MayJunMean+*R*)	291.94	9	311.10	0.0354
δ(MayJun28dHigh) μ(MayJun28dHigh+*R*)	299.29	7	314.00	0.0083
δ(MayJun28dHigh+*R*) μ(OctAprMean+*R*)	295.52	9	314.69	0.0059
δ(MayJunMean+*R*) μ(OctAprMean+*R*)	297.16	9	316.33	0.0026
δ(OctAprMean+*R*) μ(MayJun28dHigh+*R*)	297.46	9	316.62	0.0022
δ(JulSepMean+*R*) μ(MayJun28dHigh+*R*)	297.78	9	316.94	0.0019
δ(MayJunMean) μ(MayJun28dHigh+*R*)	302.96	7	317.67	0.0013

[1] = Models included all δ and μ combinations of null effects (.), random effects (*R*), and TIHMs/flow metrics (with and without *R*).
[2] = Likelihood (–2[log-likelihood]) was estimated for each model.
[3] = Higher numbers of parameters indicate increased model complexity.
[4] = Top ten models were ranked (best to worst) by Akaike's information criterion (AIC$_c$) and include the AIC$_c$ weight (w_i); lower AIC$_c$ and higher w_i indicate a better model.

DISCUSSION

Comparison to Isleta and San Acacia Reach Analyses

Hydrologic and Geomorphic Conditions

Hydrologic conditions in the Angostura Reach were generally characterized by higher flow magnitudes relative to the Isleta and San Acacia Reaches (Figure 45). Peak flow metrics for the Angostura Reach (28d high May–June) were higher than the Isleta Reach for 25 of 27 years in the study period (1993–2019) and higher than the San Acacia Reach for all years. Similarly, mean flow metrics corresponding to the principal life-stages of the Rio Grande Silvery Minnow in the Angostura Reach were higher than the Isleta and San Acacia Reaches in 76 and 80 of 81 of the selected mean flow metrics 1993–2019, respectively. Low flow metrics (7d low July–September, October–April) were highest in the Angostura Reach for all years – no zero values were recorded for the Angostura Reach and only one value was <100 cfs (2013, July–September). Due to spatial variability in flow conditions within each Reach, the flow metrics recorded for analyses herein are a simplistic representation of hydrologic conditions at the reach-scale that do not fully capture the complexities of flow at the subreach-scale. In the Angostura Reach, spatial variability in discharges was expected to be relatively low, especially in comparison to the San Acacia Reach, however, several notable diversions and outfalls exist in the Angostura Reach (e.g., ABCWUA diversion and outfall, AMAFCA outfalls) and can cause varied flows to occur. Consistency in the selection of flow metrics in Process-Linkage Reports I–III provided a basis for comparison of seasonal, biologically relevant hydrologic conditions between the Angostura, Isleta, and San Acacia Reaches of the Middle Rio Grande.

Analysis of channel morphology over time in the Angostura Reach indicated geomorphic trends most similar to those observed in the Isleta Reach. Notably, channel planforms in these reaches were predominantly characterized as transitioning to the migrating stages (M) of the planform evolution model (Massong et al., 2010) over time, representing incision and narrowing withing a predominantly single-threaded channel. The presence of aggrading stages (A) was exclusively observed in the San Acacia Reach. Within the Angostura Reach, the Bernalillo and Montaño subreaches showed similar trends in channel evolution over the study period (1962–2012), however, the upstream-most subreach B1 showed the largest magnitude of channel incision within the study area. Additionally, the Angostura Reach showed the highest density of jetty jacks compared to downstream reaches. Reduced sediment supply into the Angostura Reach following completion of Cochiti Dam (1973) has caused channel incision and floodplain disconnection over time (Richard and Julien, 2003); our results show that most changes to the channel occurred between 1972 and 1992. In locations where degradation has diminished (2012), local processes such as bed coarsening and bank armoring (e.g., vegetation, jetty jacks) potentially restrict channel evolution through migrating stages (i.e., state of arrested degradation; M5). At the reach-scale, lateral migration rates appear to be low, however, local bed and bank erosion may occur in locations where prevailing processes (e.g., bank armoring) are not limiting channel evolution.

Habitat Conditions

Flow-habitat curves generated for the Angostura Reach were similar to those from the Isleta and San Acacia Reaches for initial survey years 1962–1972 but showed reach-specific trends for 1992–2012 (Figure 46). For the initial survey years (1962 and 1972), flow-habitat curves for all reaches showed progressive increases in habitat availability with increasing discharge with peaks at about 5000 cfs and similar peak magnitudes of habitat availability (4–5 x 10^6 ft^2 day/mi). The Angostura Reach showed higher habitat availability at lower discharges (<2000–3000 cfs) likely due to differences in modeling methods between reaches (see Data and Methods – Bankfull Discharge). For recent survey years (1992, 2002, and 2012), flow-habitat curves differed between reaches. The Angostura and Isleta Reaches showed similar decreases in habitat availability, however, the Angostura Reach showed much more gradual increases in habitat availability with increasing discharge, whereas the Isleta Reach showed sharp increases – these sharp increases corresponded to estimated bankfull discharge values, which were impacted by differences in hydraulic modeling methods between reaches. Whereas the Angostura and Isleta Reaches were generally characterized by reduced habitat availability for each principal life-stage

across discharges (Q<5000 cfs), the San Acacia Reach maintained relatively high habitat availability within the same range of flows. These findings were influenced by reach-specific modeling methods and do not fully represent field conditions specific to inundation thresholds across reaches.

Analysis on the effects of geomorphic change on TIHMs generally showed decreases to habitat availability across reaches for all life-stage periods over time. However, several notable exceptions were observed for larval TIHMs. In the Angostura Reach, an increase in larval TIHMs occurred between 2002 and 2012 for low and moderate flow scenarios (Figure 32). The Montaño Reach showed local channel aggradation between 2002 and 2012 (Figure 4), which might explain this increase, yet the percentage contribution of that subreach remained diminished relative to upstream (i.e., Bernalillo). In the Isleta Reach, larval TIHMs remained elevated for both high flow scenarios over time (1962–2012). Similarly, in the San Acacia Reach, larval TIHMs remained elevated for both high flow scenarios and one moderate flow scenario over time (1962–2012). For the Isleta and San Acacia Reaches, these results suggest that sufficiently high flows still produce substantial increases in larval habitat availability relative to historical datasets — in comparison, the Angostura Reach appears to have sustained greater losses to larval habitat availability at high flows.

Time integrated habitat metrics (TIHMs) also differed across reaches during the study period (Figure 47). Larval TIHMs for the Angostura Reach were greater than the Isleta and San Acacia Reaches in 24 and 12 of 27 years compared during the study period (1993–2019), respectively. Juvenile and Adult TIHMs in the Angostura Reach were greater than the Isleta Reach for all years; juvenile and adult TIHMs in the Angostura Reach were greater than the San Acacia Reach in 21 and 18 of 27 years for these life-stage periods, respectively. Greater TIHMs values in the Angostura Reach were attributed to two main factors: (1) tendency for higher flow magnitudes to occur in this reach and (2) pronounced differences in flow-habitat curves across reaches 1992–2012 (Figure 46). Mean, peak, and low flow metrics showed flows in the Angostura Reach tended to be highest across the seasonal hydrologic periods selected. Additionally, flow-habitat curves for the Angostura Reach showed marginally higher levels of habitat availability at low–moderate discharges compared to the Isleta Reach (1992–2012) and San Acacia Reach (2012). These factors interacted and accumulated over the duration of the life-stage periods to produce relatively higher TIHMs in the Angostura Reach in some years, however, the Angostura Reach showed considerably less variation in TIHMs than downstream reaches (Figure 47). Larval TIHMs for the Isleta Reach exceeded the Angostura Reach only during exceptionally high flow years (1994, 1997, and 2005). Larval TIHMs for the Angostura Reach tended to be higher than the San Acacia Reach during exceptionally low flow years (e.g., 2001–2003 and 2011–2016); larval TIHMs tended to be greater in the Angostura Reach when differences in mean flow May–June between reaches were greater than 50%. These results suggest that when increases in habitat availability associated with overbanking are not produced (i.e., relatively low spring runoff magnitude and duration), habitat availability, as measured by TIHMs, will be greater upstream, due to decreases in flow magnitude along a downstream gradient. The tendency for higher flows to occur in the Angostura Reach contributed to relatively higher juvenile and adult TIHMs, however, the San Acacia Reach occasionally produced higher values.

Although the TIHM analyses showed that the Angostura Reach tended to have greater habitat availability than downstream reaches, it is unclear if the TIHMs provided equivalent comparisons of field conditions across reaches 1993–2019. As noted for the Isleta Reach, derivation of flow-habitat curves did not fully capture spatial variations in floodplain inundation and it is possible that this modeling artifact resulted in underestimates of habitat availability, and subsequently TIHMs, for discharges just below estimated bankfull discharges (Mortensen et al., 2020). Therefore, the reach comparisons provided herein should be interpreted cautiously in relation to population responses of Rio Grande Silvery Minnow across reaches. Nonetheless, the flow-habitat curves derived for these analyses show how differences can produce pronounced differences in habitat availability related to seasonal flow variations.

Locations of habitat formation also appeared to differ within and across reaches. For example, larval habitat in the Angostura Reach was primarily contributed by the Bernalillo subreach — habitat mapping showed that at moderate to high flows (>1,500 cfs) habitat in this subreach occurred in inundated secondary channels. This results contrasts habitat mapping results from the Isleta or San Acacia Reaches, which tended to show that increases in habitat availability were associated with floodplain inundation at moderate to high flows (>3,000 cfs). Differences in hydrologic conditions and habitat availability between reaches illustrated the role geomorphology serves as a mediator between flow and hydraulic conditions in the Middle Rio Grande.

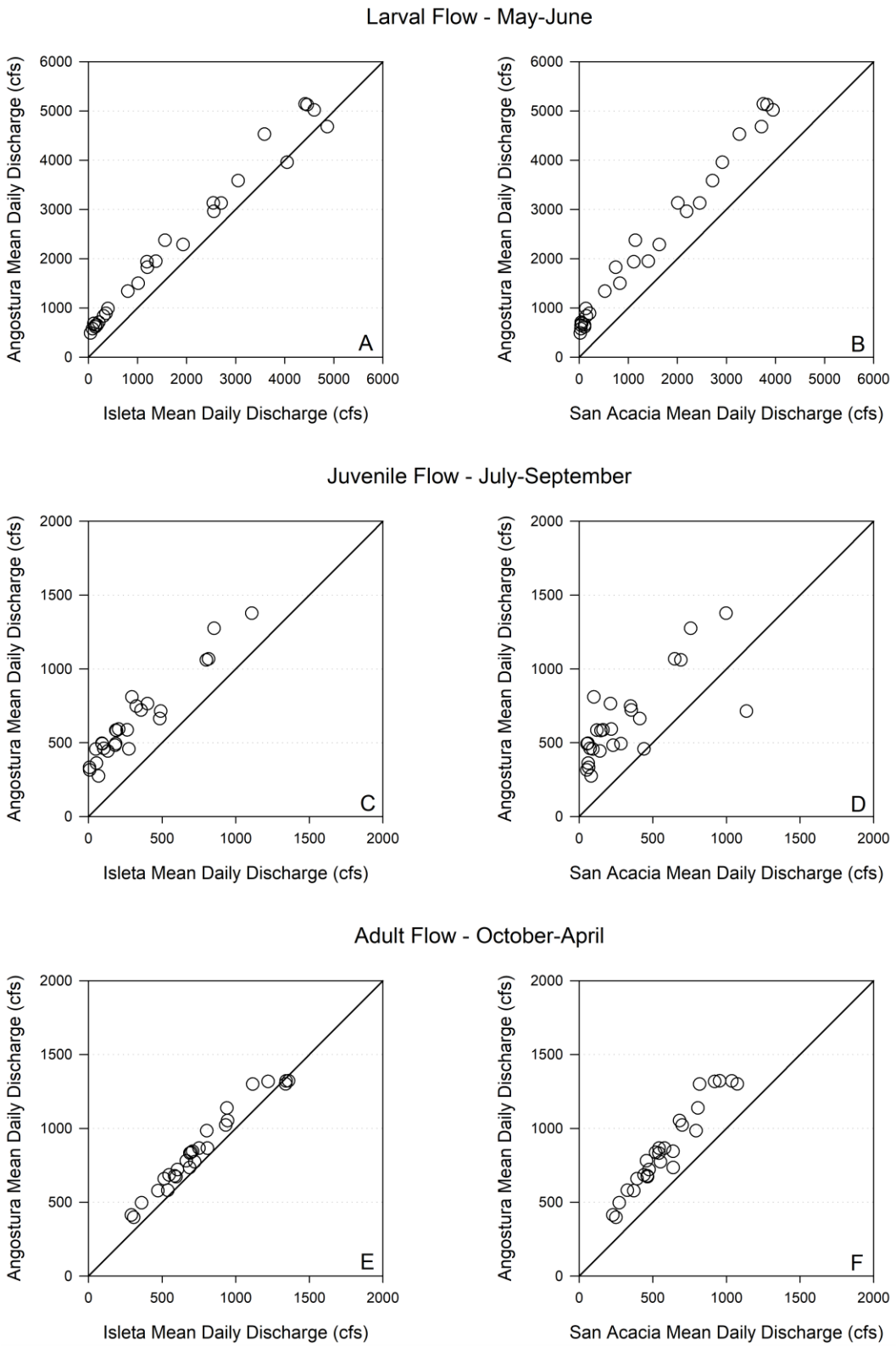


Figure 45. Comparison of flow metrics (mean daily discharge) between the Angostura Reach (y-axes) and the Isleta (1993–2019) and San Acacia Reaches (x-axes; 1993–2021).

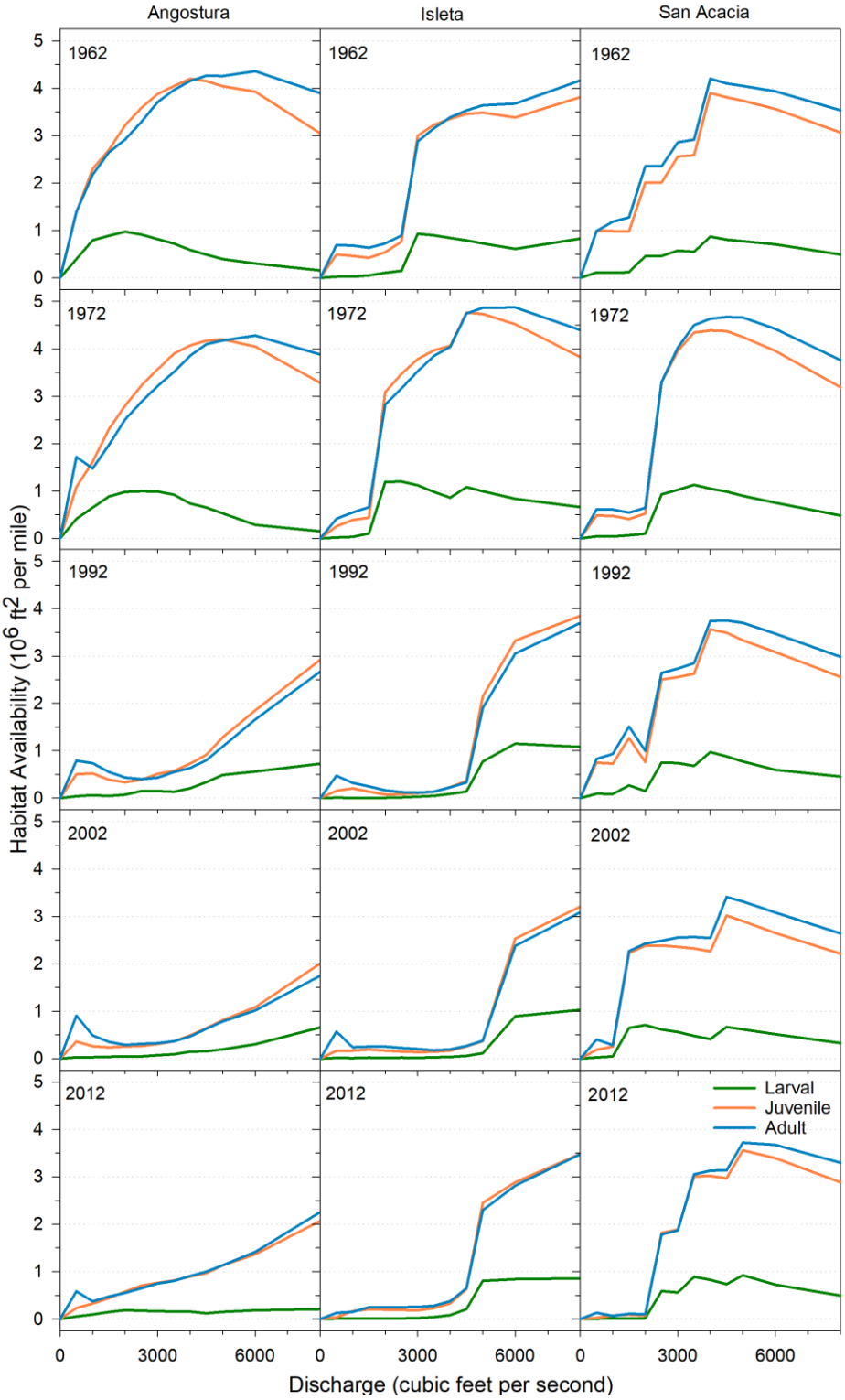


Figure 46. Comparison of flow-habitat curves for Middle Rio Grande reaches: Angostura, Isleta, and San Acacia. Curves are shown through time top to bottom (1962–2012). Line colors represent the primary life-stages of the Rio Grande Silvery Minnow. Habitat availability was normalized by reach length.

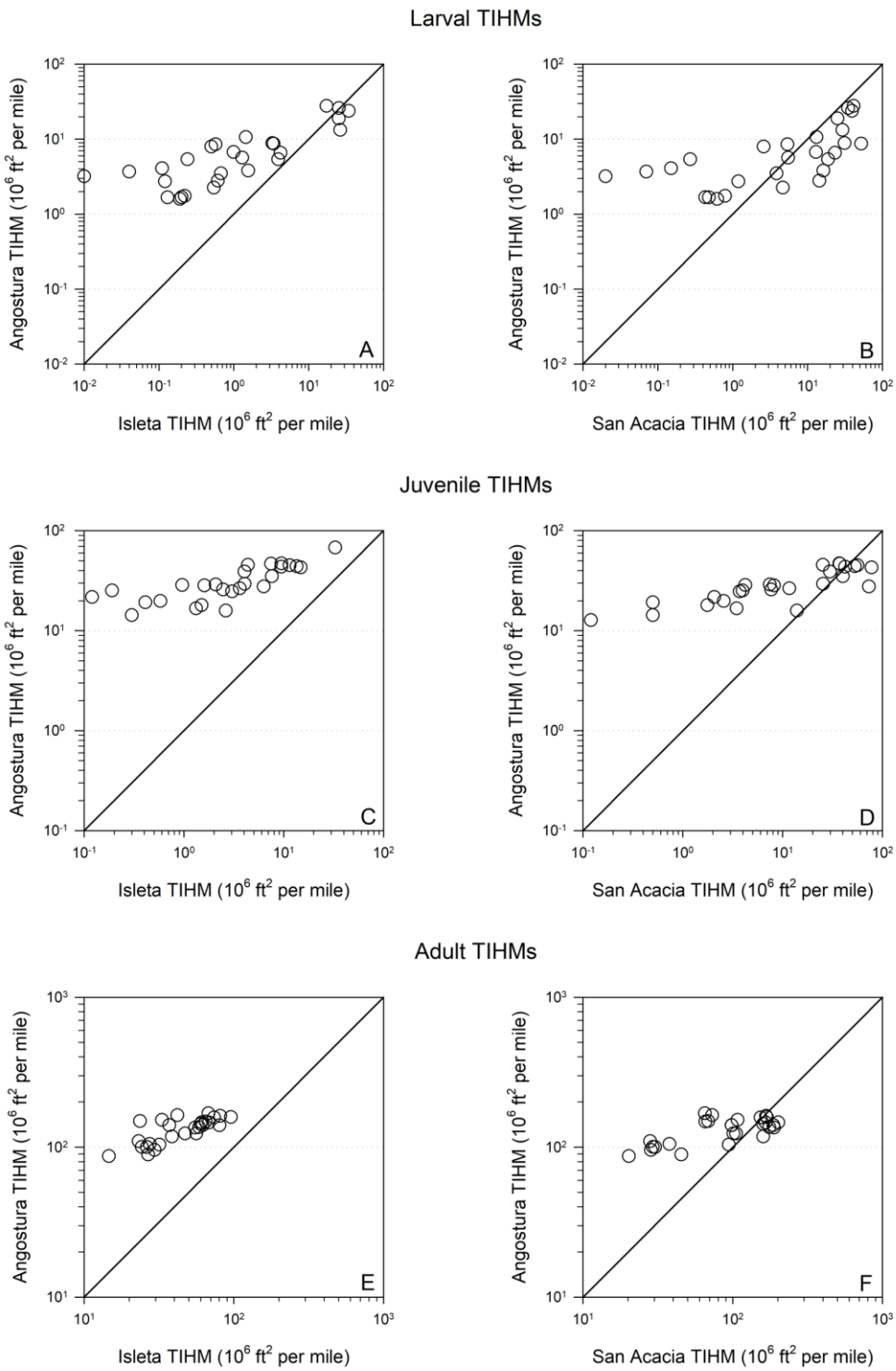


Figure 47. Comparison of Time Integrated Habitat Metrics (TIHMs) between the Angostura Reach (y-axes) and the Isleta (1993–2019) and San Acacia Reaches (x-axes; 1993–2021).

Long-term Ecological Relationships

Density and occurrence estimates of the Rio Grande Silvery Minnow (i.e., October sampling) at the reach-scale differed between the Angostura, Isleta, and San Acacia Reaches. Densities of the species were generally highest in the San Acacia Reach over time but varied between the Angostura and Isleta Reaches – no clear trend between reaches was observed for occurrence probabilities (Figure 48). Higher densities in the San Acacia Reach may be attributed to higher larval habitat availability (Figure 47b) and the propensity for downstream drift of eggs and larvae from upstream reaches. Although differences in density and occurrence estimates were evident across reaches, these down-scaled values generally followed the same trends over time across reaches (Figure 49) and were therefore similar to range-wide patterns for the Middle Rio Grande (Dudley et al., 2022).

Comparison of changes in occurrence and density of the Rio Grande Silvery Minnow during October (1993–2021) with habitat and flow metrics revealed several strong ecological relationships. Elevated and prolonged flows during the spawning/rearing season (i.e., primarily May–June) were closely related to the increased occurrence and density of the Rio Grande Silvery Minnow. Similarly, we found that higher availability estimates of larval fish habitat, during May and June, were associated with an increased occurrence and density of this species throughout the study period.

Analysis of TIHMs and flow metrics (i.e., ecological relationships) indicated that flow metrics consistently explained more variation in the population of Rio Grande Silvery Minnow, as compared to TIHMs, across reaches and across years (1993–2021). TIHMs did not fully capture spatial and temporal variations of floodplain inundation and habitat formation. This factor was first noted during analysis of the Isleta Reach, where flow-habitat relationships showed sharp increases in habitat availability over a small range of discharges. These sharp increases corresponded to the exceedance of bankfull discharges, which were approximated via hydraulic modeling methods (Mortensen et al., 2020), thus, small differences in flows often had notable impacts on the predicted amount of larval habitat — these modeled responses do not fully reflect the subtle complexities of floodplain inundation processes (e.g., more gradual responses and/or lower thresholds) across a long and varied reach over the study period (1993–2019). In the San Acacia Reach, perched and semi-perched channel conditions presented similar modeling challenges (i.e., need to estimate bankfull discharges via hydraulic modeling) that may have contributed to inaccuracies in modeling inundation of low-lying areas. The Angostura Reach showed more gradual increases to habitat availability with increasing flows, which might be attributed to differences in the modeling methods implemented in this reach (i.e., bankfull discharges were not estimated due to channel incision), however, it is unclear if these results provide an accurate representation of flow-habitat relationships. These analytical factors might partially explain why flow metrics, as compared to TIHMs, were consistently more predictive of the increased occurrence and density of this species over time.

Across reaches (Angostura–Isleta–San Acacia), the highest-ranking flow metrics corresponded to peak discharge over a 28-day period during May–June (MayJun28dHigh). These flow metrics consistently outranked flow metrics corresponding to mean discharge during the same period (May–June). Although these two flow metrics were correlated, the higher ranking of the 28-day peak flow metric could be related to the biological responses. The 28-day flow duration was selected based on the approximate developmental period for larval Rio Grande Silvery Minnows (Platania, 2000). Elevated flows occurring over this duration are expected to correspond to the persistence of nursery habitats that provide the bio-physical conditions required for sufficient growth and development of larval fishes, the most sensitive and vulnerable life-stage. These results indicate that multiple characteristics of spring discharge (e.g., duration and timing) should be considered with respect to other pertinent flow characteristics (e.g., magnitude).

Modeling distinct population responses (occurrence vs. density), using both habitat and flow metrics, provided valuable insights into long-term population trends for the Rio Grande Silvery Minnow. Our analyses indicated that the magnitude and duration of peak spring flows were most predictive of reach-wide increases in the occurrence and density of this species over time (i.e., increases in larval habitat due to elevated and prolonged spring flows consistently explained higher density and occurrence of the species over time). In contrast, habitat and flow metrics for juveniles or adults were not as predictive of these reach-wide increases, further highlighting the importance of increased habitat created by spring flows for larval fish. Over the past two decades, similar relationships between spring flows and range-wide increases in this species across years have been documented (Dudley et al., 2022). Similarly,

higher numbers of young Rio Grande Silvery Minnows, collected in isolated pools during episodic river-drying events from June to October (2009–2015), were associated with elevated mean May discharge over time (Archdeacon, 2016). Additionally, augmentation rates were not incorporated into our analyses although stocking can be substantial (Archdeacon, 2023) – in spite of stocking effects, peak spring flows remained the most indicative of population dynamics over time, further supporting the implications for managing spring flows to achieve range-wide increases in the population. Flow metrics corresponding to the juvenile and adult life-stages held little predictive power in the Angostura and Isleta Reaches and only showed marginally better performance in the San Acacia Reach (Mortensen et al., 2020; 2023). Despite our key findings regarding spring flow conditions, flow and habitat conditions for juveniles and adults should not be entirely neglected as adverse impacts caused by low flow periods are well documented (Archdeacon and Reale, 2020).

Prolonged and elevated spring flows result in overbank flooding of vegetated areas, formation of inundated habitats within the river channel, and creation of shoreline pools and backwaters. These shallow low-velocity habitats, which typically increase in number and extent during spring runoff, are essential for the successful recruitment of larvae for many freshwater fishes throughout the world (Welcomme, 1979; Junk et al., 1989; Matthews, 1998). In the absence of adequate spring flows (e.g., during extended droughts), however, pelagic-spawning cyprinids appear to be particularly susceptible to recruitment failure (Perkin et al., 2019). It is likely that similar processes are affecting the survival and recruitment of native fishes in the Middle Rio Grande, including early life stages of the Rio Grande Silvery Minnow (Pease et al., 2006; Turner et al., 2010; Hoagstrom and Turner, 2013; Dudley et al., 2022).

Analyses of long-term ecological relationships indicated challenges associated with metrics of habitat availability. In particular, TIHMs were highly influenced by the derived flow-habitat relationships and were thereby affected by uncertainty associated with the modeling of hydraulically suitable habitat conditions. This effect, which was also noted for the Isleta and San Acacia Reaches, likely impacted our ability to elucidate strong relationships between TIHMs and fish densities (as compared to flow metrics) due to the complexities of floodplain inundation and habitat formation processes that are difficult to model accurately given data and modeling constraints. Also, TIHMs were calculated based on fixed dates for each principal life-stage period (e.g., May–June for the larval life-stage), which might not precisely reflect key life-stage transitions in the fish population across years (i.e., fish respond to environmental stimuli to initiate life-stage transitions not fixed monthly periods). However, the use of fixed monthly periods provided a tractable, systematic basis to compare TIHMs by life-stage across years and reaches (i.e., Angostura vs. Isleta vs. San Acacia). In addition to the TIHMs and flow metrics analyzed herein at the reach-scale, additional factors likely impacted population dynamics of the species during the study period (e.g., thermal regimes, downstream drift/dispersal), however, such factors were not explicitly included for these analyses. Although we note these analytical considerations in our analyses, valuable insights were gained regarding key environmental drivers of Rio Grande Silvery Minnow population dynamics over time. Specifically, we found that increased availability of larval habitat during spring (May–June) was associated with increased densities of the Rio Grande Silvery Minnow across years.

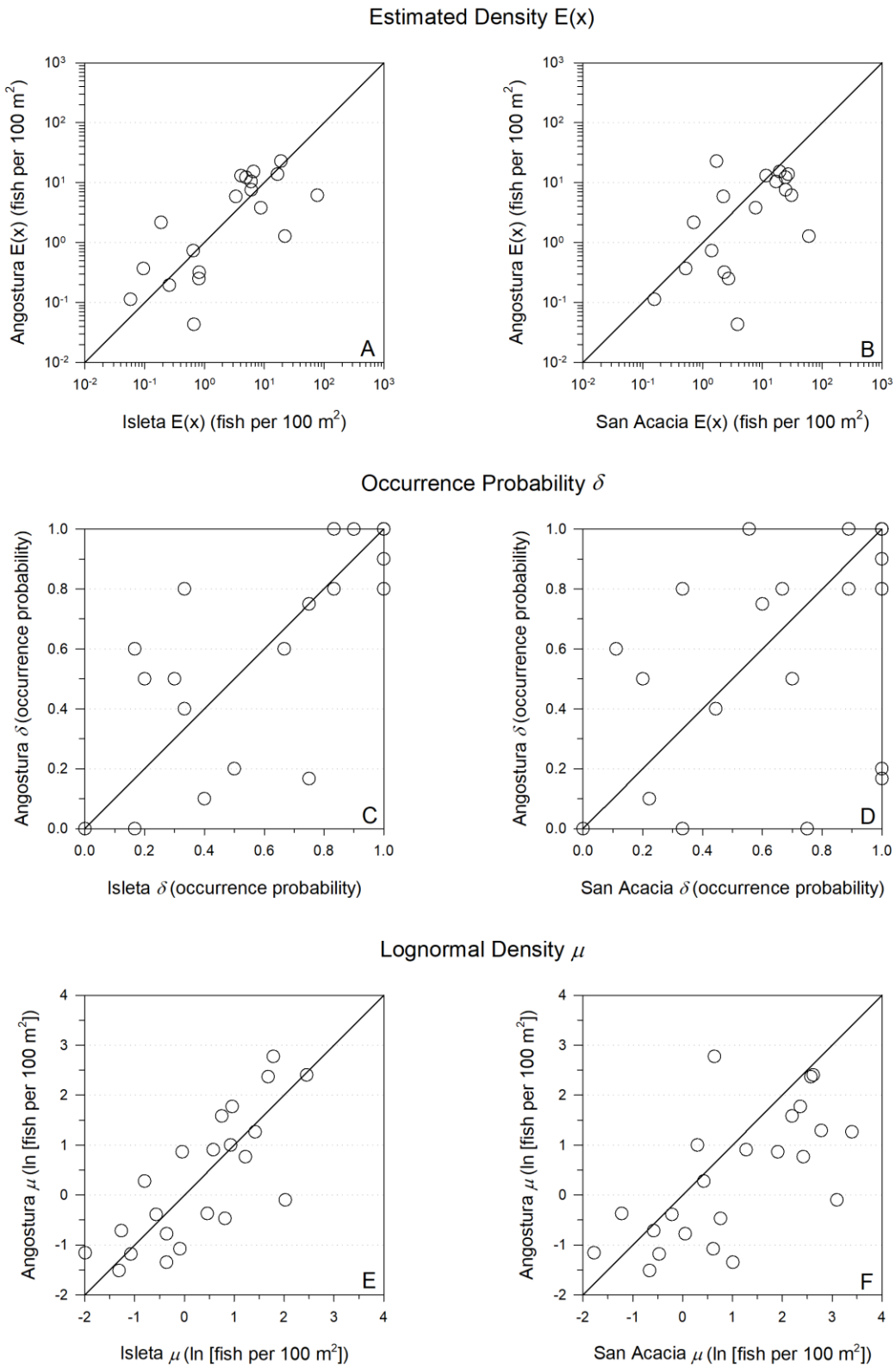


Figure 48. Comparison of estimated densities E(x), occurrence probabilities (δ), and lognormal densities (μ) of the Rio Grande Silvery Minnow between the Angostura Reach (y-axes) and the Isleta and San Acacia Reaches (x-axes)

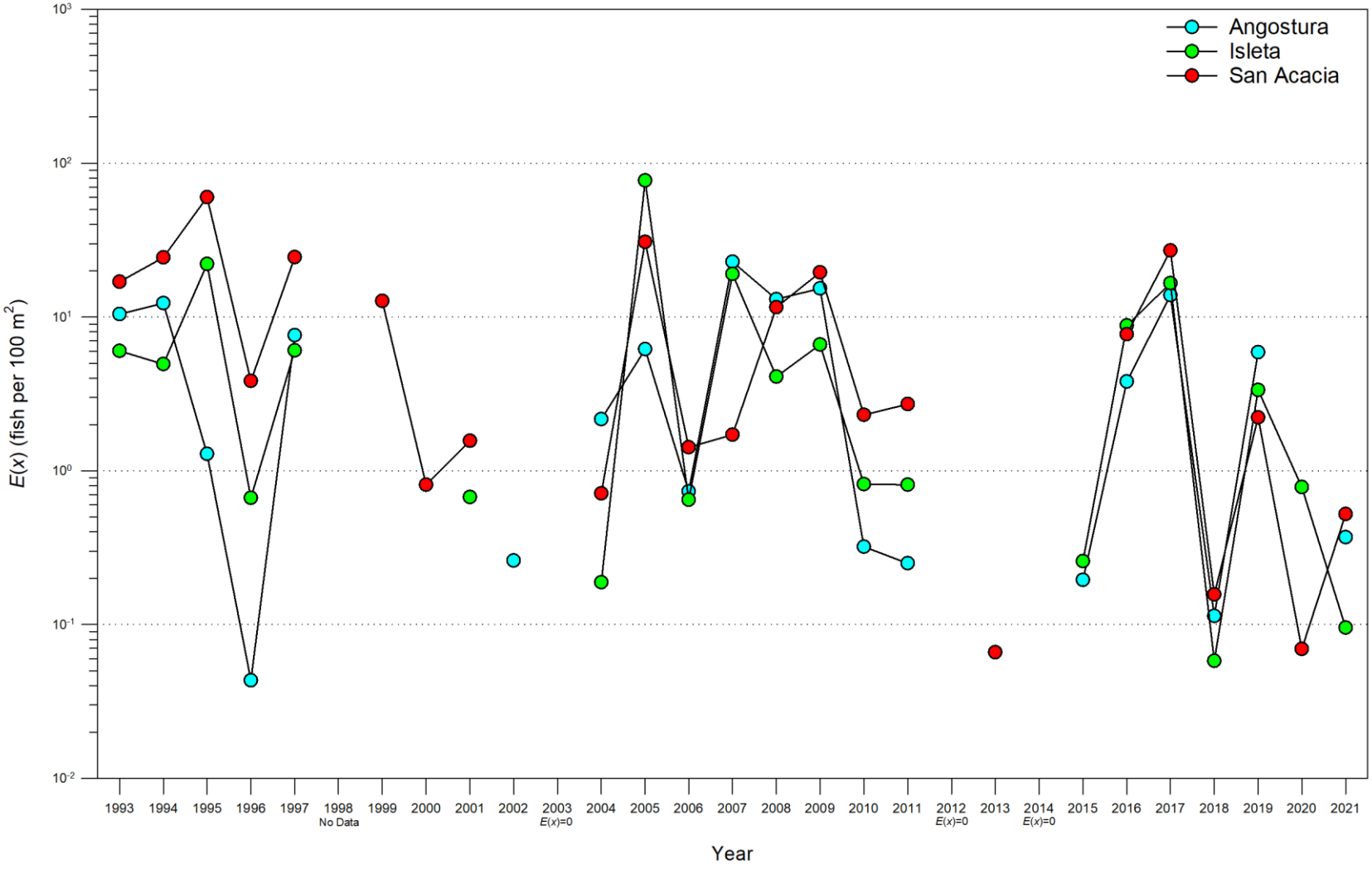


Figure 49. Densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data) from each reach (Angostura, Isleta, San Acacia) across years (1993–2021).

Key Process-Linkages in the Middle Rio Grande

Process-Linkage 1: Main Channel Habitat Complexity and Availability

Main channel complexity and habitat availability was previously identified as a secondary process-linkage in this study (Isleta and San Acacia Reaches) and was considered less important than floodplain connectivity in increasing the availability of larval habitat (Figure 50; Mortensen et al., 2020; 2023). Due to the prevalence of channel incision and reduced floodplain connectivity in the Angostura Reach, flows were typically confined to the primary and secondary channels, and therefore, main channel habitat complexity was the primary process-linkage identified for this reach. Main channel complexity is determined by the channel-floodplain morphology, which is closely linked to flow and sediment regimes, channel evolution processes, river engineering, and riparian vegetation. Habitat availability is determined by the interaction between streamflow and channel morphology. Main channel habitat complexity is important for meeting habitat requirements of the Rio Grande Silvery Minnow across the range of stream discharges.

Geomorphic trends in the Angostura Reach revealed notable trends in main channel habitat complexity and availability. Habitat mapping results showed the formation of hydraulically suitable habitats in secondary channels with increasing discharge (1,500, 3,000, and 5,000 cfs). These habitat conditions occurred predominantly in the Bernalillo subreach, the most incised subreach in the study area. Due to the high rates of channel incision in this subreach, it was not expected to contribute substantially to habitat availability, particularly for the larval life-stage. Interestingly, the Bernalillo subreach consistently provided higher relative contributions to larval TIHMs over time (1993–2021; Figure 27). These results show how secondary channels can contribute to habitat availability when flows are elevated, even in incised sections of the Middle Rio Grande. Although connected floodplains would be expected to provide larger increases in habitat availability with increasing flows, the finding that main channel complexity (e.g., multiple channels) as a driver of larval habitat availability carries implications for habitat restoration in the Middle Rio Grande.

Unlike floodplain habitats, which typically only persist seasonally (given prolonged overbank flows during spring), main channel habitats are important year-round and for all life-stages of the Rio Grande Silvery Minnow. For each of the life-stage, main channel habitat complexity and availability influences rates of survival. During May–June, if peak flows do not cause overbanking, spawning is restricted to main channel habitats (Dudley et al., 2019). In this case, main channel habitat complexity influences the capacity to retain eggs and larvae (i.e., surface transient storage or dead zones) and the availability of nursery habitats, which support growth and survival of larvae. During July–September, juveniles depend on sufficient flows to maintain the availability of suitable habitats (e.g., water temperatures, dissolved oxygen). During October–April, juveniles and adults depend on the availability and stability of overwinter habitats to survive until spring. This study considered how changes to instream habitat complexity and availability impact the population of the Rio Grande Silvery Minnow.

Although main channel complexity has been impacted over time, population monitoring trends suggest that instream habitat availability is adequate to support the Rio Grande Silvery Minnow during juvenile and adult life-stages *given sufficient flows are available during these periods*. Specifically, density and occurrence of the Rio Grande Silvery Minnow were negatively affected by extended low flow periods (e.g., number of days Q<200 cfs; Dudley et al., 2022). Prolonged low flow periods are indicative of increased likelihood of river drying, which is known to rapidly deteriorate habitat conditions and cause high mortality rates due to reduced habitat availability, water quality degradation, and terrestrial predation (Cave and Smith, 1999; Archdeacon, 2016; Archdeacon and Reale, 2020; Van Horn et al., 2022). Additionally, ecological relationships between environmental variables and population parameters investigated in this study showed that both TIHMs and flow metrics corresponding to juvenile and adult life-stage periods were relatively weak predictors of abundance and occurrence of the Rio Grande Silvery Minnow across reaches (as compared to spring flows). These results suggest that increases in habitat availability during July–April do not strongly correspond to increased density or occurrence of the Rio Grande Silvery Minnow. While these results suggest that main channel habitat availability does not strongly influence population dynamics, the discharge variations for the juvenile and adult life-stages during the study period were relatively low and it is possible that larger variation could indicate stronger effects. It is also important to note that under the contemporary channel morphology, main channel

habitat complexity alone does not appear to sufficiently provide the habitats needed for egg retention and larval development (i.e., nursery habitats), these habitats appear to be more closely linked with floodplain connectivity and inundation (Process-Linkage 3), which in comparison, can increase larval habitat availability (i.e., larval TIHM) by orders of magnitude. This is supported by the tendency for low density and occurrence of the Rio Grande Silvery Minnow to occur in years when spring runoff did not cause considerable overbanking (Dudley et al., 2022). It is unclear if increased instream habitat complexity and availability at low to moderate discharges (<3,000 cfs) would be sufficient for retaining and rearing larval Rio Grande Silvery Minnows during years with low water availability in the Middle Rio Grande (i.e., unable to provide overbanking flows during May–June). Accordingly, there is growing interest in developing and managing side channels and instream bars and islands to substitute for floodplain connectivity in incised river channels (McComas et al., 2022; Holste et al., 2023).

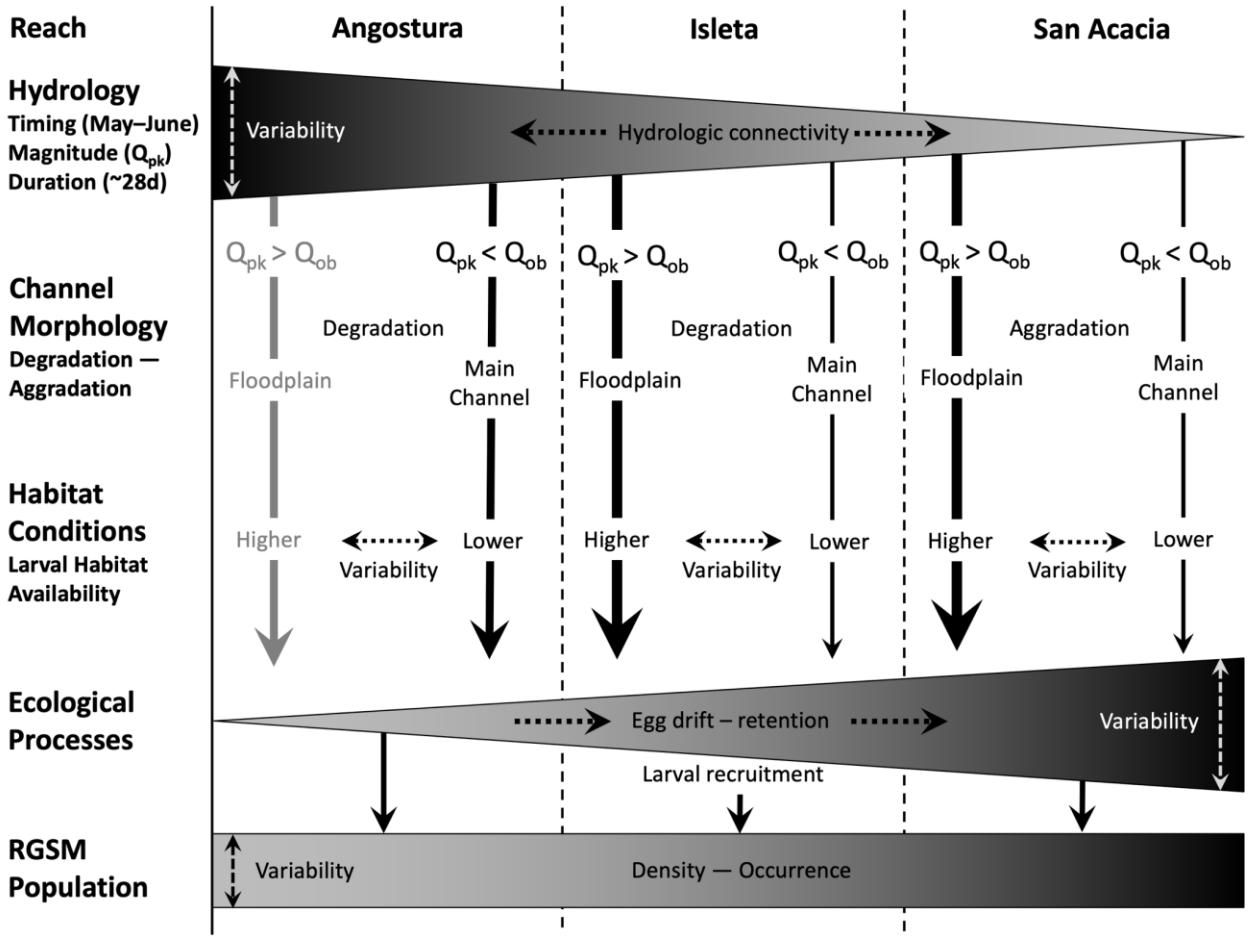


Figure 50. Conceptual diagram of key process-linkages in the Middle Rio Grande. Arrows represent linkages between seasonal hydrologic conditions, reach-specific channel morphology and habitat conditions, ecological processes, and reach-scale trends in the population of Rio Grande Silvery Minnow. Line weights of vertical arrows represent the relative importance of peak flow magnitude (Q_{pk}) on larval habitat availability based on reach-specific flow thresholds (overbanking discharge, Q_{ob}). Gradient and thickness (hydrology, ecological processes, and RGSM population) represent spatial trends in the Middle Rio Grande — flow magnitude decreases downstream, net drift of eggs and larvae increases downstream, and fish densities increase downstream.

Process-Linkage 2: Hydrologic Connectivity

Hydrologic connectivity was identified as a secondary process-linkage in the Middle Rio Grande (Figure 50). Here, hydrologic connectivity refers to the connectivity of lotic habitats within the Angostura Reach (e.g., between subreaches) as well as the connectivity of the Angostura Reach to downstream reaches (e.g., Isleta and San Acacia Reaches).

Sustained connectivity of lotic habitats within reaches is important to the survival of juvenile and adult Rio Grande Silvery Minnows. Extreme low flows are not as common in the Angostura Reach relative to downstream reaches, however, it is possible for this reach to experience intermittent flows, as recorded during the summer of 2022 (McKenna, 2023). Although TIHMs and flow metrics July–September were not shown to be strong predictors of Rio Grande Silvery Minnow (i.e., density and occurrence in October sampling), prolonged low flow periods and flow intermittency cause catastrophic fish mortality events (Archdeacon and Reale, 2020). Recent studies indicate that Rio Grande Silvery Minnow, do not engage in synchronized, population-level movements to perennially flowing areas or suitable low flow refugia, rather these fishes become trapped in proximal, short-lived habitats (e.g., isolated pools) where water quality rapidly degrades (Archdeacon et al., 2021, 2022; Van Horn et al., 2022). Furthermore, management of flow recession rates does not appear to be an effective strategy to mitigate stranding of fish during flow intermittence (Archdeacon et al., 2022). As such, strategies to maintain flowing conditions should be prioritized to reduce mortality of the species during low flow periods.

The Angostura Reach is located near the upstream boundary of the Middle Rio Grande and the suite of hydrological, morphological, and ecological conditions that occur in this reach carry implications for downstream reaches. In particular, the interaction of spawning activity by the Rio Grande Silvery Minnow and hydrological conditions in the Angostura Reach influences the magnitude of ichthyofaunal drift (i.e., downstream dispersal of eggs and larvae) into the Isleta and San Acacia Reaches (Figure 50). For example, given presence of the Rio Grande Silvery Minnow and prolonged floodplain connectivity in upstream reaches during the spawning period, it is likely that the magnitude of downstream ichthyofaunal drift will be reduced due to relatively high rates of upstream retention. Conversely, given presence of the species and the absence of floodplain connectivity in upstream reaches, it is likely that the magnitude of downstream ichthyofaunal drift will be increased. Downstream reaches (i.e., Isleta and San Acacia Reaches) tend to contain higher densities of juvenile Rio Grande Silvery Minnows (1993–2021) despite intensive augmentation efforts upstream — this pattern is likely explained by the cumulative downstream dispersal of eggs, larvae, and young-of-the-year (Dudley et al., 2022). The interactions between floodplain connectivity and propagule retention in the Angostura Reach are likely to be a continual challenge for management of Rio Grande Silvery Minnow in this reach due to the propensity for downstream displacement of offspring. Additionally, the Middle Rio Grande extends 36.2 km (22.5 mi) upstream to Cochiti Dam (i.e., Cochiti Reach) – Rio Grande Silvery Minnow is suspected to be extirpated from the Cochiti Reach. In contrast to the Isleta and San Acacia Reaches, the lack of a spawning population upstream of the Angostura Reach precludes drift into the reach. Therefore, strategies that increase propagule retention and the availability of nursery habitats (e.g., flow management, habitat restoration) will be important for local recruitment of Rio Grande Silvery Minnow in the Angostura Reach.

Hydrologic connectivity in the Middle Rio Grande is affected by dams, which carry implications for geomorphology and ecology. Cochiti Dam drastically reduced the volume of sediment supplied to the Angostura Reach, which has led to channel incision and reduced floodplain connectivity (Process-Linkage 3). As discussed herein, these geomorphic impacts are strongly related to habitat conditions needed by the Rio Grande Silvery Minnow. Sediment transport is important for morpho-dynamics and habitat maintenance and creation processes. Additionally, dams restrict upstream dispersal of fishes. The Angostura Reach, due to its upstream location, low floodplain connectivity and habitat fragmentation, is especially susceptible to net downstream losses of offspring (i.e., eggs, larvae, and juveniles) and absent management actions to counteract this process (i.e., augmentation and fish passages), progressive declines in the species abundance would be expected to occur over time. Rio Grande Silvery Minnow are capable of ascending fishways, and their implementation has been successful on similar rivers (e.g., Arkansas River), including upstream recolonization by pelagic-broadcast spawning fish (Archdeacon and Remshardt, 2012; Pennock et al., 2017). Reach-specific differences in habitat conditions revealed by this study show how process-linkages can vary spatially with implications for future conservation planning.

Process-Linkage 3: Floodplain Connectivity and Inundation

Floodplain connectivity and inundation did not appear to be prevalent in the Angostura Reach (1993–2021), yet this process-linkage remains highly important to the ecology of Rio Grande Silvery Minnow. This linkage has been the primary finding for downstream reaches (i.e., Isleta and San Acacia_ and shows the complex interactions among hydrologic and geomorphic processes in the Middle Rio Grande (Figure 50; Mortensen et al., 2020; 2023). Differences in floodplain connectivity and habitat formation across reaches were apparent in habitat mapping results (Figure 51).Floodplain connectivity and inundation are controlled by hydrologic and geomorphic factors functioning over multiple spatiotemporal scales. Changes to the primary drivers of channel evolution processes in the Middle Rio Grande, flow and sediment regimes, in combination with river engineering efforts, land use changes, and riparian vegetation have largely determined the present morphology of the channel and floodplain (Massong et al., 2006; Petrakis et al., 2017). In the Angostura Reach, the channel has generally incised and stabilized in migrating planform stages, although lateral mobility in this reach appears to be low (Massong et al., 2006; 2010). This process has largely disconnected the floodplain and reduced the availability of shallow, low velocity habitats, particularly those that were historically available during snowmelt runoff April–June, though some localized flooding appears to occur around 5,000 cfs (Anderson et al., 2023). The San Acacia Reach, which often had lower flow magnitudes, sometimes produced greater larval habitat availability estimates (i.e., TIHMs) relative to the Angostura Reach, which appeared to correspond to the maintenance of floodplain connectivity by channel aggradation (Mortensen et al., 2023).

Despite greater channel incision and lower floodplain connectivity in the Angostura Reach relative to downstream reaches, our analyses of long-term ecological relationships in this study demonstrated the importance of flow and habitat conditions that are representative of the ecological processes that occur seasonally (i.e., peak flows May–June and larval habitat availability). Within the Angostura Reach, the Bernalillo subreach showed higher percentage contributions to reach-scale larval habitat availability (i.e., larval TIHMs), relative to the Montaño subreach, over time (1993–2021). Habitat mapping results showed that the increased habitat availability in the Bernalillo subreach primarily occurs in secondary channels that are activated at flows >1,500 cfs (Radobenko et al., 2023). While secondary channels are also present in the Montaño subreach, these channels appear to convey more flow across moderate to high discharges (1,500–5,000 cfs), which excludes these areas as hydraulically suitable habitat (i.e., these side channels are relatively deep with swift current; Anderson et al., 2023). Recent research on the Middle Rio Grande has targeted instream features and side channel dynamics and carries implication for the design of sustainable habitat features (McComas et al., 2022; Holste et al., 2023). The timing, duration, and frequency of floodplain and side channel inundation are important for providing adequate habitat conditions for the survival of early life-stages of the species.

The life history of the Rio Grande Silvery Minnow is important for understanding the role of floodplain connectivity in supporting reproduction and recruitment of this species in the Middle Rio Grande. The Rio Grande Silvery Minnow is a short-lived species – the majority of the population lives about one year (Horwitz et al., 2018). Consequently, abundance in the wild is largely determined by the seasonal availability of shallow, low velocity habitats each year. The early life-stages of this species (i.e., eggs and larvae) are highly susceptible to downstream displacement (Dudley and Platania, 2007). In fragmented and reservoir bound river systems such as the Middle Rio Grande, local recruitment likely depends on the availability of habitat features that increase hydraulic residence times and the retention of early life-stages in upstream reaches (i.e., eggs and larvae are not displaced into Elephant Butte Reservoir or entrained at water diversions). Floodplain morphology and characteristics (e.g., vegetation) decrease water velocities and entrain eggs and larvae (Valdez et al., 2019, 2021), reducing the downstream flux of propagules into unfavorable habitats (e.g., E. Butte Reservoir), and increase the availability of shallow water depths, which provide spawning and nursery habitats for the Rio Grande Silvery Minnow (Pease et al., 2006; Magaña, 2012; Gonzales et al., 2014; Valdez et al., 2019, 2021). The duration of nursery habitat availability (ca. one month) is also important for achieving rapid and sufficient growth and development through early life-stages. Seasonal floodplain inundation (May–June) increases the availability of these specific habitat conditions in the Middle Rio Grande.

This study demonstrated the significance of floodplain connectivity and inundation to the wild population of the Rio Grande Silvery Minnow through ecological relationships between environmental

variables (e.g., TIHMs and flow metrics) and population monitoring parameters (e.g., estimated density and probability of occurrence). Both habitat and flow metrics during the larval life-stage period (May–June), which correspond to seasonal peak flows (i.e., spring runoff), were consistently among the most reliable predictors of increased density and occurrence of the Rio Grande Silvery Minnow across reaches of the Middle Rio Grande (Tables 9–11). Recreating floodplain habitat features (e.g., inset floodplains, compound channels), restoring floodplain connectivity, and floodplain creation processes (e.g., lateral channel mobility) are becoming increasingly common goals of ecological recovery programs of large river systems. Recent research efforts on the Sacramento River, CA, the Trinity River, CA, and the Missouri River basin have identified the functional role floodplains in providing habitat conditions for numerous threatened and endangered species (The Nature Conservancy et al., 2008; Trinity River Restoration Program, 2009; Jacobson et al., 2014). As in the Middle Rio Grande, these large river systems are also challenged by modified flow and sediment regimes, long-standing development, water and land use, and changing climatic conditions. Overcoming these challenges while maintaining water resources for human benefit (e.g., water supply and flood protection) will be needed to meet the habitat requirements of threatened and endangered species and achieve long-term conservation of biodiversity in these environments.

Geomorphic Controls on Process-Linkages

The key process-linkages identified for the Middle Rio Grande illustrate the dynamic and complex interactions among streamflow, channel-floodplain morphology, and the availability of hydraulically suitable habitat. The Angostura Reach showed spatially uniform channel evolution trends during the study period (i.e., M stages) that impact key process-linkages and the creation of habitat conditions in this reach. For example, the incised channel morphology appeared to limit increases in larval habitat availability for low to moderate flow scenarios 1992–2012 as compared to 1962–1972, with only slight increases observed between moderate and high flow scenarios in 2012 (Figure 32). These results suggest that contemporary channel morphology controls relationships between discharge and habitat availability such that only exceptionally prolonged and high magnitude flows produce pronounced increases in habitat availability (i.e., TIHMs).

Planform evolution, as described by the MRG model, contains processes not fully observed in the Angostura Reach. While all subreaches have followed the migrating (M) reach trajectory (Massong et al., 2010), progression to later migrating reach stages (M6–M8) was not observed. The absence of these model stages could be attributed to three factors, (1) channel incision has not been great enough to undermine banks and cause bank erosion, (2) bank erosion is restricted by bank armoring (e.g., jetty jacks and riparian vegetation), and/or (3) the spatial scale analyzed failed to capture areas of localized channel migration. The first factor is related to Lane's Balance (Figure 11), which describes how channel characteristics adjust to changes in flow and sediment discharge. The second factor is controlled by natural and manmade factors such as vegetation and channelization (Figure 13), respectively. In the Angostura Reach, the high density of jetty jacks has influenced channel evolution in this reach by stabilizing banks and reducing lateral mobility. Therefore, it seems unlikely that the Angostura Reach will undergo considerable change to its channel morphology under prevailing environmental conditions. Additionally, levees, urban and agricultural infrastructure, and municipal land use adjacent to the river in this reach restricts the space available for lateral movements. Constraints on lateral channel adjustments render many assumptions of classic channel evolution models to be inappropriate, such that natural channel responses to transport imbalances (i.e., Lane's Balance) do not follow the predicted trajectory of lateral mobility and widening following channel incision (Smith et al., 2008; Booth and Fischenich, 2015). Recent studies in channel evolution have explicitly incorporated the role of external factors on channel processes as well as the tendency for channel evolution to progress in a cyclical pattern with the potential for 'dead-end,' 'short-circuits,' and skipped stages to occur during this cycle (Cluer and Thorne, 2013; Booth and Fischenich, 2015; Castro and Thorne, 2019; Johnson et al., 2020) – the Angostura and Isleta Reaches appear to largely be in a state of arrested degradation. Laterally active channels are expected to lead to improved habitat and ecosystem benefits as channels evolve towards a quasi-undisturbed state (Cluer and Thorne, 2013). For the Angostura Reach, the lack of lateral mobility is likely to restrict the potential for natural processes to improve hydraulic habitat conditions. However, there is some capacity within the established bank lines for channel evolution to occur (e.g., bar and island formation), which

might allow for the formation of shallow, low-velocity habitats – techniques to promote the formation of bars, islands, and secondary channels are gaining interest (McComas et al., 2022; Holste et al., 2023). Additionally, reduced sediment supply into the Angostura Reach (i.e., entrapment in Cochiti Reservoir) is likely to limit natural processes from recovering floodplain connectivity and main channel habitat complexity (Process-Linkages 1 and 3). Overall, the insights gained from the synthesis of channel evolution models and habitat analyses might be used to inform planning of habitat restoration efforts of the Middle Rio Grande.

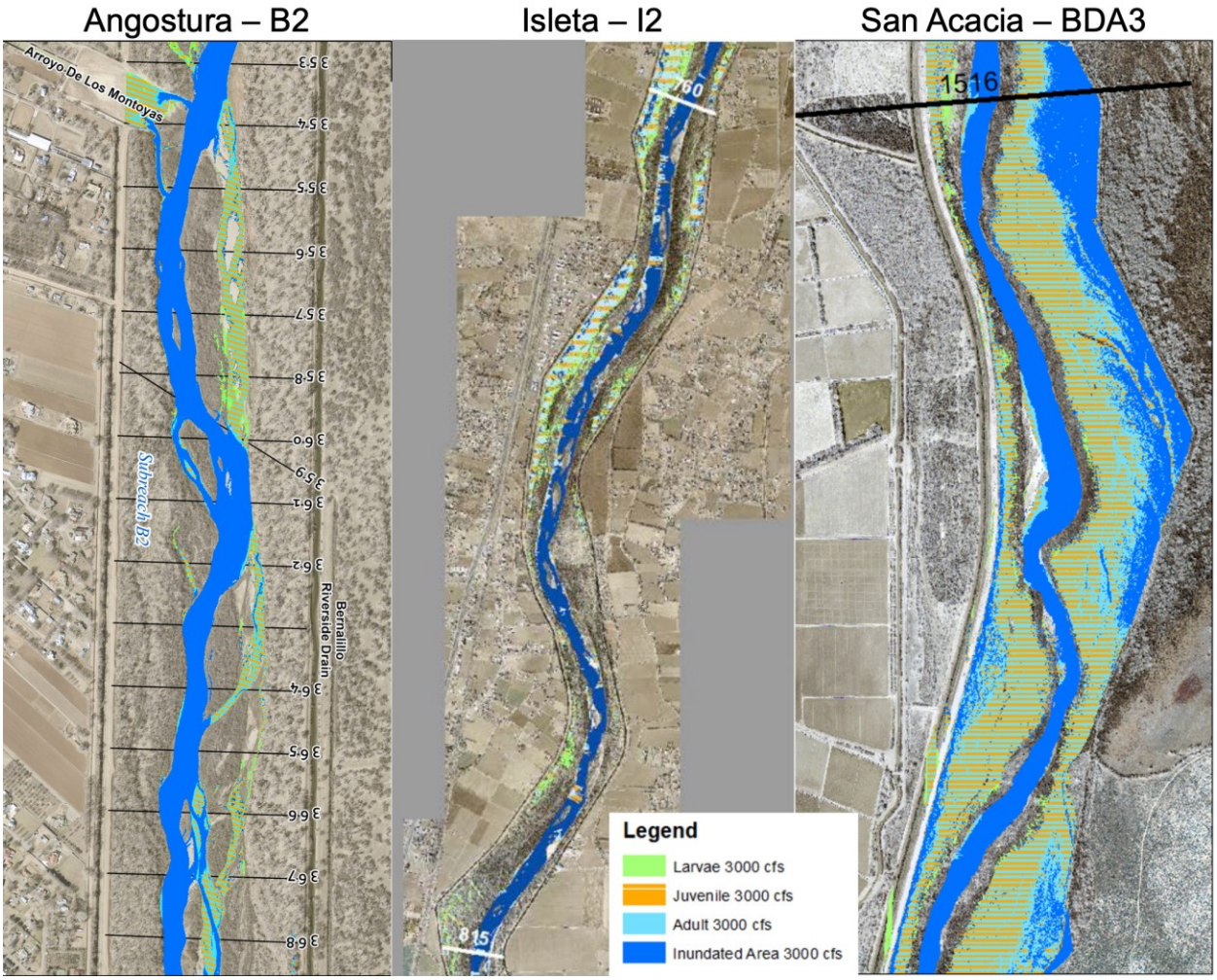


Figure 51. Comparison of habitat mapping results at 3000 cfs from three reaches of the Middle Rio Grande: Angostura, Isleta, and San Acacia. Differences in floodplain connectivity and habitat formation are apparent. Note scale differs between panels. Images from Radobenko et al., 2023 (Angostura); Mortensen et al., 2020 (Isleta); Schied et al., 2022 (San Acacia).

Analytical Considerations and Data Collection Recommendations

Channel Morphology

The greatest limitations encountered during this study involved the spatial and temporal resolution of channel surveys. While channel surveys broadly captured changes in floodplain and channel elevations over time, two key limitations were identified across reaches: (1) limited accuracy of main channel bed elevations associated with low flows (<500 cfs) and cross-section spacing (~500 ft), and (2) low temporal resolution of channel surveys (ten-year survey interval).

Bed elevations were estimated indirectly using methods that did not capture finer scale variations in elevations of the main channel bed surface. Bed elevations and channel profiles were estimated using an assumed trapezoidal cross-section, and water surface elevation and approximate discharge at time of survey (Varyu, 2013). The resulting cross-sections therefore lack a detailed depiction of the channel surfaces occurring beneath the water surface. This was especially problematic when deriving habitat availability estimates for low flow periods (<500 cfs), which occur frequently in the study area, because effectively no direct measurements of channel elevations were available. Accordingly, inferences regarding habitat availability during low flow periods should be interpreted cautiously. Moderate to high flows were less likely to be impacted by these limitations due to the measurement of floodplain elevations. Due to this limitation, it is recommended that future channel surveys occur during low flow periods to better characterize bed elevations within the main channel. Similarly, cross-section spacing (~500 ft) did not capture variations in channel morphology between aggdeg lines, providing a simplified representation of longitudinal variations in channel morphology.

Channel surveys occurred at approximately ten-year intervals, which introduced uncertainty regarding channel conditions during interim periods. While flow-habitat relationships showed relatively low magnitude changes 1992–2012 relative to 1972–1992 in relation to changing channel morphology, the lack of interim data prompted the assumption of minimal temporal changes to channel-floodplain elevations between surveys. While this assumption seems reasonable based on the available data, we were unable to assess these associated impacts due to the lack of detailed main channel bed elevations. Rangeline data, which contains surveyed measurements of channel bed elevations could be used to retrospectively assess these assumptions. As such, it is recommended that channel surveys continue either at closer fixed intervals or to correspond to defined flow targets. Channel surveys could be timed to target specific flow conditions, such as low, moderate, and high flows, which would improve characterization of channel-floodplain elevations and inundation patterns. Monitoring of channel conditions before and after high flows would provide data to characterize how periods with heightened stream power (i.e., peak flows), influence morphological dynamics. Under the assumption that streamflow driven morphological changes are relatively minimal during low to moderate flows, it is possible that the frequency of channel surveys could be reduced during drought periods, however, the encroachment of riparian vegetation during low flow periods (i.e., low stream power) should be considered (Massong et al., 2010). Overall, the uncertainty associated with main channel elevations, flow-inundation patterns, and temporal changes in channel morphology warrant intensive monitoring be continued.

Hydraulic modeling methods were also a source of inaccuracy for estimating flow-habitat relationships. One-dimensional hydraulic modeling approaches (i.e., HEC-RAS) are primarily suited for estimating water surface elevations and are limited in their ability to calculate fine-resolution hydraulic parameters (i.e., depth and velocity) and lateral water distributions. For our purposes, the basic data available for the study area (e.g., lack of detailed channel bathymetry, coarse cross-section spacing) likely did not support the potential gain of using a two-dimensional hydraulic model. The width-slice method was implemented in this study to attempt to overcome these limitations, however, it is unclear how well the hydraulic modeling results reflect conditions in the field. Furthermore, estimating bankfull discharges was also problematic in reaches with perched or semi-perched channel conditions due to limitation of the hydraulic modeling methods (Mortensen et al., 2023). This effect contributed to notable differences between Angostura and Isleta Reach results. TIHMs were consistently higher in the Angostura Reach despite greater floodplain connectivity in the Isleta Reach – higher flows in the Angostura Reach were not expected to cause such pronounced differences in habitat availability. It is recommended for data collection efforts to monitor inundation patterns during spring runoff to improve accuracy of flow-habitat relationships through ground-truthing of bankfull discharge thresholds.

Determining bankfull discharges for past datasets retroactively will likely continue to be problematic, however, improvements could be made for future data collections. Likewise, two-dimensional models could be developed for selected areas (e.g., subreaches) to better characterize flow-habitat relationships across a broad range of discharges (e.g., 50–7,500 cfs) and refine bankfull discharge estimates – prior studies that determined flow-habitat relationships using two-dimensional modeling were limited to a relatively low range of discharges and a sparse selection of sites (Bovee et al., 2008). Given the data available for the Middle Rio Grande and the objectives of this study, HEC-RAS served as a tractable, well-established approach to assess long-term trends in channel morphology and habitat availability. Nonetheless, further studies seeking to determine quantitative relationships between streamflow and habitat availability should consider the implementation of two-dimensional modeling approaches as logistically feasible.

Hydrology

Discharge measurements obtained from USGS gaging stations had the highest temporal resolution of the selected datasets (i.e., mean daily discharge). Spatial resolution was generally suitable for characterizing hydrological conditions at the reach-scale. However, spatial variation in flows within and across reaches occurs frequently. While these variations were generally considered to be negligible in terms of estimating habitat availability metrics (i.e., TIHMS) in the Angostura Reach, characterization of several flow conditions could be informative for ecological impacts.

Low flow periods can produce intermittent flow conditions. These conditions occur most frequently in the Isleta and San Acacia Reaches, however, intermittency extended upstream to the Angostura Reach in 2022. Low flows and intermittent flows adversely affect the Rio Grandy Silvery Minnow by increasing mortality during these periods, yet relating drying severity (e.g., extent and duration) to population parameters remains challenging due to a lack of long-term monitoring data or established relationships between streamflow and drying extent. Therefore, analyses herein have relied on low flow metrics as a surrogate for river drying, which likely do not fully capture the range of drying impacts. The RiverEyes monitoring program has collected data on river drying since approximately 1996 (McKenna, 2023); this information has yet to be synthesized into values suitable to aid direct assessment of river drying and the Rio Grande Silvery Minnow (i.e., as covariates for modeling long-term ecological relationships). It is recommended that relationships between streamflow and river drying be determined to provide quantitative estimates of drying severity (e.g., river mile-days). While low flow impacts were demonstrated by this study, primarily in the Isleta and San Acacia Reaches, their relative impact on overall population dynamics was considerably lower than flow conditions associated with spring runoff.

Flow conditions can also vary considerably within reaches. The Angostura and Isleta Reach analyses included a single discharge record for each reach due to data availability constraints, whereas San Acacia Reach analyses incorporated discharge measurements for each subreach due to greater data availability in that reach — this allowed for more accurate spatial representation of habitat conditions as captured by TIHMs. Spatial variation in discharges appeared relatively minimal in the Angostura Reach, however, notable diversions and outfalls contribute to variations in flows (e.g., ABCWUA intake and outfall, irrigation returns). In the Isleta Reach, flow conditions can vary considerably especially during low flow periods that cause intermittency. Future studies should continue to account for spatial variation in flow conditions to the extent feasible given established gaging stations and data availability.

Rio Grande Silvery Minnow

The Rio Grande Silvery Minnow Population Monitoring Program provided systematic estimates of population parameters for the past several decades (1993–2021). The distribution of sampling sites provided robust estimates of Rio Grande Silvery Minnow population parameters (i.e., density and occurrence) for the Middle Rio Grande and despite fewer sampling sites available for reach-scale analyses, downscaled population parameters were obtained. Based on trends observed across reaches, it is unlikely that additional benefits would be gained from further downscaling of population parameters (i.e., subreach) even if sampling sites were added to facilitate reducing spatial scales. Range-wide environmental factors, primarily spring runoff magnitude and duration, were shown to have much stronger influence on population responses across reaches than localized habitat conditions. Additionally, the

Population Monitoring Program has recently added sites to April and October sampling events for a total of 30 sites (10 per reach) to strengthen reach-scale estimates (Dudley et al., 2022). Therefore, modifications to this long-term monitoring study are not currently recommended. However, additional insights might be gained through integration of reach-scale results to the Middle Rio Grande (i.e., range-wide).

The covariates evaluated in our analyses of ecological relationships (i.e., TIHMs and flow metrics) provided a quantitative basis to evaluate the influence of variations in environmental conditions on the fish population, yet as previously noted, these metrics did not fully capture spatial and temporal variations of discharge or the full suite of biological factors that might have affected population dynamics during the study period (1993–2021). For example, low flow metrics provided a surrogate for negative interactions related to river drying, declining water quality (e.g., high water temperatures, low dissolved oxygen), and heightened biotic factors (e.g., stress, predation), however, they did not explicitly account for these impacts quantitatively. Similarly, the delineation of key life-stage and seasonal periods contained some overlap across life-stages and hydrologic conditions — for example, protracted spring runoff (i.e., occurring before or after May–June) could have carried over into Oct–April and July–September flow metrics. Also, covariates did not account for antecedent conditions in the fish population (e.g., fish density the previous year) or the potential effects of management interventions (e.g., stocking, rescue). While it is likely that additional factors are influencing population dynamics over time, the selected metrics were chosen to focus on the interactions amongst streamflow, habitat availability, and the population of Rio Grande Silvery Minnow.

The lack of population monitoring data prior to 1993 precluded inferences regarding ecological relationships related to changes in hydrology and channel morphology 1962–1992. Changes in channel morphology and flow-habitat curves were less dramatic for the period 1992–2012 as compared to 1962–1992; the absence of fish population data for this period precludes quantitative analysis of ecological relationships. Anecdotally, Rio Grande Silvery Minnow were abundant and widely distributed in the Middle Rio Grande 1962–1972, however, this cannot be verified, therefore, any inferences regarding the population prior to 1993 are merely speculative. Regrettably, it does not seem possible to retrospectively evaluate ecological relationships quantitatively prior to the initiation of the Rio Grande Silvery Minnow Population Monitoring Program in 1993.

Conditions during the larval life-stage of Rio Grande Silvery Minnow (May–June) were shown to be the strongest predictor of distribution and abundance of the species in October. Despite the importance of this sensitive life phase, key knowledge gaps exist regarding larval fish in the Middle Rio Grande. Specifically, it would be valuable to gather more information on the timing, distribution, and persistence of larval Rio Grande Silvery Minnow (July–September) in habitats of the Middle Rio Grande. Studies have documented the timing and presence of larvae in natural and constructed floodplain habitats (Valdez et al., 2019; 2021). Further studies are recommended to improve our understanding of larval fish dynamics in the Middle Rio Grande.

Channel-Habitat Evolution Model

This study observed several limitations of the planform evolution model developed by Massong et al., (2010). For example, cross-sectional data, which was not incorporated into the MRG model, showed channel incision occurring prior to transition to stage M4 (i.e., first stage of the migrating reach). This was particularly evident in the Angostura and Isleta Reaches because they have incised during the study period (Mortensen et al., 2020). Therefore, the addition of designations to stages 2 and 3 (e.g., M2 and M3) might be warranted to better describe this degradational trend. Similarly, renaming of the migrating (M) stages as degrading (D) could be more descriptive, particularly for reaches that do not appear to exhibit widespread lateral adjustments. Additionally, the roles of external factors were not incorporated into the original MRG model. In the Angostura and Isleta Reaches, high densities of jetty jacks were installed prior to 1960 and these structures in combination with riparian vegetation protect banks from erosion, which can prevent channel evolution from progressing to stage M6, the stage that signifies the initiation of lateral channel migration. In the San Acacia Reach, reservoir levels impacted channel evolution processes in the downstream-most subreach (Holste, 2015; Mortensen et al., 2023) – the Elephant Butte subreach experienced a period of aggradation when reservoir levels were elevated (1992–2002), which was followed by a period of degradation when reservoir levels fell (2002–2012). Also, sediment plugs that formed in the San Acacia Reach during the study period did not progress to the point of developing a new channel alignment via natural processes (stage A6), rather pilot channels were dredged, thereby controlling channel evolution processes by returning flow to the existing channel alignment. Finally, it remains unclear how future multi-year drought periods will affect morpho-dynamic trends. In the planform evolution model, the transition to stage 3 is preceded by several consecutive low flow years, which allows riparian vegetation to stabilize channel features, leading to channel narrowing and potentially increased rates of channel incision. If such processes recur within the narrowed and incised channel, this could represent a ‘short-circuit’ in channel evolution at described by the stream evolution model (SEM; Cluer and Thorne, 2013). Continued monitoring and surveying of channel morphology across flow conditions is recommended to further our understanding of ongoing geomorphic processes and resulting evolutionary trajectories for the Middle Rio Grande.

Implications for River Management Practices

Habitat Restoration

Actions proposed to alleviate threats to the Rio Grande Silvery Minnow include the restoration and protection of habitats in the Middle Rio Grande (USFWS, 2010). Ongoing habitat restoration in the Middle Rio Grande is largely focused on creating floodplain habitats to be inundated during years with low to moderate spring discharges (USBR, 2012). Our findings from habitat analyses across reaches carry implications for the restoration of aquatic habitats in the Middle Rio Grande.

The flow-habitat relationships derived by this study can be used to provide a rough estimate of the spatial extent of habitat that has been lost relative to historical conditions. For example, using a discharge of 2000 cfs, the difference between 1962–1972 and 2012 larval habitat availability values in the Angostura Reach was approximately 0.80 x 10^6 ft^2 per mile – this is equivalent to a width of about 75 ft on both sides of the river for the length of the Angostura Reach. This estimate is provided to help visualize the spatial scale of habitat loss that has occurred along this section of river as well as the potential scale of habitat restoration needed for incised channels, however, it should be interpreted cautiously as habitat availability estimates were not intended to be precise quantifiers of habitat area. From 1985 to 2008, the width of the channel in this reach was estimated to have decreased on average by 98 ± 92 ft (30 ± 28 m; Swanson et al., 2011); which suggests that habitat loss might have been driven by channel narrowing during this period. Additionally, habitat maps generated for this study (2012) can identify low-lying areas as prospective locations for increasing floodplain connectivity. For example, in the Montaño subreach, numerous locations between the I-40 and Bridge Blvd. crossings, near the Tijeras arroyo confluence, and near the I-25 bridge crossing show inundated areas at 3,000–5,000 cfs as well. These areas might be conducive to increase local floodplain connectivity due to their low elevations.

The results of this study indicate that habitat restoration projects should consider location within the system (upstream versus downstream) and local channel processes (e.g., channel aggradation versus degradation). Given the reproductive ecology of the Rio Grande Silvery Minnow, a species that produces eggs and larvae that are highly susceptible to downstream displacement, restoring floodplain connectivity in upstream reaches, like the Angostura Reach, is expected to be beneficial because such actions should contribute to higher rates of propagule retention in upstream reaches. Yackulic et al., (2022) also suggested that restoration of larval habitat in the Angostura Reach carried higher potential to be an effective management action when compared to other actions across reaches. Relative to the San Acacia Reach, which appeared to maintain floodplain connectivity at moderate to high flows in this study, the Angostura Reach has largely lost floodplain connectivity. The maintenance of floodplain connectivity via natural processes in the San Acacia Reach suggests that construction of low-velocity habitats (e.g., side channel, floodplains, backwaters, embayments) should be prioritized for upstream reaches that have experienced reductions or losses to floodplain connectivity due to channel incision (e.g., Angostura and Isleta Reaches). Side channels in the Bernalillo subreach appeared to contribute to habitat availability at moderate to high flows, suggesting that construction of these features could be beneficial when designed to provide low-velocity habitat at low to moderate discharges (1000–3000 cfs). Habitat restoration in the Angostura and Isleta reaches is likely to be impacted by the legacy of jetty-jacks in these reaches; habitat restoration projects should investigate interactions among jetty-jacks, morpho-dynamics, and habitat restoration to inform future management. Facilitating bank erosion is expected to provide beneficial ecological impacts but might require infrastructure setbacks to allow sufficient space for meander migration and flood zones (Florsheim et al., 2008; Biron et al., 2014). Incorporating geomorphic context (i.e., morphological processes and infrastructure impacts) into planning of habitat restoration activities is needed to overcome limitations associated with current restoration approaches (e.g., narrow focus and short-term planning; Harris et al. 2023). Overall, restoration activities need to consider prevailing hydrologic and morpho-dynamic trends (e.g., flow frequencies, aggradation vs. degradation, incised vs. perched channels), site-specific factors within the prospective location, and potential trajectories of various management actions (i.e., how restoration sites are expected to change over time).

Our understanding of habitat restoration projects is currently limited with respect to their long-term functionality and their individual and cumulative impacts to the Rio Grande Silvery Minnow population at reach and range-wide spatial scales. This study was not intended to assess the effects of past or ongoing habitat restoration efforts, however, improving our understanding of the efficacy of habitat restoration in

the Middle Rio Grande will require targeted monitoring and data collection for use in future research efforts. Since Process-Linkage Report I, which recommended habitat restoration sites should be inventoried, a geodatabase has been compiled for the Middle Rio Grande (RioRestore, GeoSystems Analysis, Inc.). Resources like this will likely be needed for future researchers to address questions regarding the long-term efficacy of habitat restoration in the Middle Rio Grande. Additionally, utilization of restored floodplain sites for spawning and nursery habitats by the Rio Grande Silvery Minnow has been documented (Gonzales et al., 2014; Valdez et al., 2019), however, it is unclear how these restored habitats contribute to population dynamics beyond the site-scale (e.g., reach-scale, range-wide). Such assessments were outside the scope of this study and targeted research will be needed to better characterize relationships between habitat restoration activities and population responses of the Rio Grande Silvery Minnow.

The application of channel evolution models across reaches of the Middle Rio Grande and recent studies of other modified river systems suggest strategies for long-term habitat restoration of the Middle Rio Grande. For example, the stream evolution model (SEM) indicates the potential for channel evolution processes to naturally recover habitat and ecosystem benefits over time (Cluer and Thorne, 2013). This study identified potential geomorphic controls on channel evolution and habitats needed by the Rio Grande Silvery Minnow. Natural fluvial processes such as bank erosion, progressive channel migration, and meander cutoffs are related to formation of new floodplains and increased habitat complexity (Florsheim et al., 2008; Smith et al., 2008), the key process-linkages identified in this study. Research efforts targeting the ecological recovery of other heavily modified river systems in North America provide further insights into restoration of vital morpho-dynamic processes. For example, in the Sacramento River, CA, the restoration of bank erosion and progressive channel migration processes were identified as critical to the formation and preservation of off-channel habitats, the exchange of sediment between the channel and floodplain, and ultimately to the recovery and maintenance of numerous native species including fish, avian, terrestrial vertebrates, and plant species (Stillwater Sciences, 2007; The Nature Conservancy et al., 2008;). Additional research efforts, environmental organizations, and river managers support these recommendations (Florsheim et al., 2008; Smith et al., 2008; Olson et al., 2014). Due to the high density of jetty-jacks in the Angostura Reach and their effects on morpho-dynamics, large-scale efforts to remove them would likely be needed to increase lateral mobility. The complexities associated with such restoration strategies are yet to be fully understood and must consider the primary drivers of channel evolution processes, flow and sediment regimes, which may pose technical and logistical constraints to ecological recovery (Jacobson and Galat, 2006; Jacobson et al., 2009). Should restoring channel migration processes to the Middle Rio Grande be identified as a habitat management strategy in the future, its application would likely require long-term planning, large-scale collaborative efforts, and gradual implementation, however, such actions might be needed to successfully achieve recovery and sustainability of habitat and ecosystem benefits in the long-term.

Flow Management

In highly modified river systems, providing environmental flows to restore morpho-dynamic processes and ecological functions is a management strategy that has gained recognition over the past several decades (Arthington et al., 2006; Yarnell et al., 2010; 2015). Accordingly, sufficient seasonal flow conditions, particularly recruitment flows and base flows, are included as criteria in the Rio Grande Silvery Minnow Recovery Plan (USFWS, 2010).

This study demonstrated intimate linkages between seasonal and annual flow conditions at the reach-scale (e.g., Angostura Reach) and the Rio Grande Silvery Minnow population over time. The strongest ecological relationships evaluated were between increased magnitude and duration of spring flows (i.e., 28 day peak flow May–June), increased availability of shallow, low-velocity habitats (i.e., larval TIHMs), and increased recruitment of the Rio Grande Silvery Minnow (Linkage 1). The mechanisms by which elevated and prolonged spring flows contribute to successful recruitment of the Rio Grande Silvery Minnow are related to interactions between the species' life-history and the spatiotemporal availability of specific hydrodynamic conditions, particularly shallow, low-velocity habitats (i.e., larval habitats selected in this study) during the spawning period. Modifications to the river and its watershed have altered the total availability and spatiotemporal characteristics of shallow, low-velocity habitats from historical conditions such that these habitats tend to be maximized at extremely low flows (e.g., Bovee et al., 2008)

and in overbank flows (e.g., this study; Adair, 2016). The management implications for environmental flows in the Middle Rio Grande to sustain the Rio Grande Silvery Minnow suggest a focus on spring runoff conditions – given the prevailing channel morphology, overbank flows are needed to produce meaningful increases in availability of larval habitats for a sufficient duration (ca. one month) and should recur at a frequency that provides successful recruitment of the Rio Grande Silvery Minnow to mitigates substantial population declines given the relatively short lifespan (typically 1–2 years) and age-class structure of this population (>95% of individuals are age-0 [autumn] or age-1 [spring]). Recent water management efforts have demonstrated the efficacy of managing spring runoff to produce positive population responses (Valdez et al., 2019). The tendency for higher magnitude flows to occur in the Angostura Reach suggests heightened potential to increase larval habitat availability in this reach by coordinating seasonal discharge increases with habitat restoration actions designed to activate during these flows. In rivers such as the Middle Rio Grande, the spring snowmelt hydrograph is increasingly recognized as an essential component of the natural flow regime that provides natural maintenance of both biotic and abiotic ecosystem processes (Yarnell et al., 2015; 2010). However, it is also acknowledged that high interannual runoff variability, frequent water scarcity (i.e., droughts), and long-standing water management practices (i.e., water uses and allocations) create a challenging atmosphere for the implementation of environmental flows in the Middle Rio Grande. Although the flow management implications of spring runoff and summer low flows on the population of the Rio Grande Silvery Minnow are evident (Archdeacon and Reale, 2020; Dudley et al., 2022), meeting these habitat requirements concurrently is likely to be challenging given the highly variable and unpredictable nature of water availability in the arid southwestern U.S. and obligations to meet current and future water supply demands. Long-term water resource planning in the Middle Rio Grande will likely require multi-faceted and innovative approaches to secure environmental flows to maintain the ecological resources of the riverine and riparian system (e.g., Richter et al., 2020).

Flow management and habitat restoration are herein described as separate management practices, however, the process-linkages demonstrated in this study between flow and channel morphology suggest that managing both flows and habitats will be needed for the recovery and long-term persistence of the Rio Grande Silvery Minnow in the Middle Rio Grande. Historically, the availability of shallow, low-velocity habitats was substantially higher across discharges in the Middle Rio Grande due to lower bankfull discharges and higher channel complexity. This study has shown that flows of sufficient magnitude and duration are needed to attain large increases in the abundance of shallow, low-velocity habitats. Given that the Rio Grande Basin is predicted to become warmer and drier over the next century (USBR, 2016; USBR et al., 2013), water resources are not currently allocated to environmental uses, and water shortages are common during drought periods, meeting the habitat requirements of the Rio Grande Silvery Minnow solely through flow management is questionable. Rather, managing the ecological resources of the Middle Rio Grande will likely require researching and developing innovative strategies to restore natural fluvial processes and sustainably rehabilitate channel-floodplain morphology to increase the availability of shallow, low-velocity habitats across a truncated range of spring runoff discharges while simultaneously maintaining human benefits (e.g., water supply, flood protection).

Species and Ecosystem Recovery

This study focused on evaluating spatiotemporal patterns of the hydraulically suitable habitat conditions needed by the Rio Grande Silvery Minnow and their potential implications for management of these habitats in the Middle Rio Grande (i.e., habitat restoration and flow management). However, additional factors likely constrain the recovery of the Rio Grande Silvery Minnow in this system. Fragmentation of riverine habitat by dams is also considered to be a principal factor in the decline of the Rio Grande Silvery Minnow (USFWS, 2010; Dudley and Platania, 2007). Fragmentation impacts the Rio Grande Silvery Minnow by inhibiting egg retention mechanisms and restricting population movement and redistribution within the river (Platania et al., 2020). These factors are important for both short-term population responses and long-term persistence. Fragmentation increases the likelihood eggs and larvae will be displaced into unsuitable habitats (Perkin et al., 2015; Perkin and Gido, 2011; Dudley and Platania, 2007). Additionally, the loss of bi-directional dispersal (i.e., most fish are unable to move upstream of dams) contributes to net downstream displacement and reduces gene flow, which is important for maintaining genetic diversity and adaptive capabilities in the wild population (Osborne et al., 2012). These

negative effects are currently mitigated through captive propagation and population augmentation programs (Osborne et al., 2006). Consequently, providing fish passages at diversion dams has been included in the Rio Grande Silvery Minnow Recovery Program and recent regulatory documents (USFWS, 2018, 2016, 2010). In addition to providing seasonally inundated floodplain habitats and reducing river drying, restoring longitudinal connectivity between reaches is anticipated to contribute to positive population responses.

Ultimately, the recovery of the Rio Grande Silvery Minnow will require not only maintaining a stable, self-sustaining population in the Middle Rio Grande, but also the reestablishment of two additional populations within the historical range of the species (USFWS, 2010). In 2008, a nonessential, experimental population of the Rio Grande Silvery Minnow was reintroduced in the Rio Grande near Big Bend, Texas, but this population is not self-sustaining (Edwards, 2017; USFWS, 2008). The successful reestablishment of additional populations of the Rio Grande Silvery Minnow will be subject to the same habitat and flow requirements evaluated in this study. Overall, it is unlikely that a ‘magic bullet’ solution exists to achieve the recovery of the Rio Grande Silvery Minnow in the Middle Rio Grande or in multiple locations in its historical range, rather, achieving ecological recovery will likely require multi-faceted, interdisciplinary approaches that restore vital interactions among hydrologic, geomorphic, and ecological processes. Restoration of key process-linkages are also expected to promote ecosystem level recovery, such as the recruitment of native riparian vegetation, the creation of habitats required by other threatened or endangered species (e.g., southwestern willow flycatcher *Empidonax trailii extimus*), and restoration of fundamental ecosystem services.

CONCLUSIONS

This study performed interdisciplinary analyses to improve understanding of the linkages among dynamic hydrologic and geomorphic processes (i.e., morpho-dynamics) and the hydraulic habitat conditions needed by the Rio Grande Silvery Minnow. The goals of this effort were consistent with recently active research programs that have implemented collaborative, interdisciplinary approaches to target ecological recovery of large, human impacted river-floodplain systems (e.g., Jacobson et al., 2014; Trinity River Restoration Program, 2009, The Nature Conservancy et al., 2008; Stillwater Sciences, 2007). We used a suite of analytical methods to integrate several long-term, systematically collected datasets that were designed to monitor and characterize hydrologic, geomorphic, and ecological trends in the Middle Rio Grande. This study furthered efforts to understand relationships between hydrogeomorphic processes and ecological dynamics occurring at the reach-scale (i.e., the Angostura Reach). We characterized relationships between discharge and habitat availability (temporally and spatially), developed a habitat metric incorporating hydrologic, geomorphic, and ecological factors over time (TIHM), evaluated long-term ecological relationships between the Rio Grande Silvery Minnow and environmental conditions (TIHMs, flow metrics), and described key linkages among morpho-dynamics processes and habitats needed by the Rio Grande Silvery Minnow.

Building on prior Process-Linkages Reports, the main findings for the Angostura Reach included:

- Key process-linkages identified for the Middle Rio Grande were: (1) floodplain connectivity and inundation, (2) hydrologic connectivity (within and among reaches), and (3) main channel habitat complexity and availability.
- Hydrologic and geomorphic conditions within the Angostura Reach showed spatially uniform trends over time (1962–2012) that were similar to those observed in the Isleta Reach (Process-Linkage Report I).
- Discharge was consistently higher in the Angostura Reach relative to downstream reaches, however, differences in TIHMs across reaches varied over time (1993–2021).
- Larval TIHMs for the Angostura Reach showed higher contributions from the Bernalillo subreach 1992–2012); habitat maps showed secondary channels created hydraulically suitable habitat when inundated.
- Densities of the Rio Grande Silvery Minnow were generally lower in the Angostura Reach than the San Acacia Reach but varied relative to the Isleta Reach. Differences in densities across reaches were attributed to differences in larval habitat availability and ecological processes (i.e., drift/dispersal).
- TIHMs and flow metrics corresponding to the larval life-stage of Rio Grande Silvery Minnow (May–June) were the most reliable long-term predictors of the species' density and occurrence across reaches, however, flow metrics explained more variation in population parameters across years.
- Data gaps and analytical considerations were identified – principally, collection of channel and floodplain elevations across flows, particularly low flows, is needed to improve modeling accuracy across reaches. Limitations to hydraulic models and habitat analyses included reduced modeling accuracy at low flows and uncertainty associated with overbanking thresholds for various channel states (e.g., incised [Angostura] vs. perched channels [San Acacia]). Targeted 2D modeling was recommended.
- Flow management and habitat restoration in the Angostura Reach will be important to aid the recovery of the Rio Grande Silvery Minnow in the Middle Rio Grande. The upstream location of this reach is important for providing spawning and nursery habitats to counter net downstream displacement of offspring. Low floodplain connectivity suggests important opportunities to increase larval habitat availability through construction of floodplains and low-velocity side channels. Higher flow magnitudes in the reach also carries potential for increasing larval habitat during spring runoff and for improving survivorship during low flow periods. However, incised channel morphology and prevalence of infrastructure, municipal land use, and bank armoring (e.g., jetty jacks, vegetation) limit potential for recovery via natural processes.

The collaboration among research institutions and river managers undertaken in this study holds promise for advancing our understanding of the Middle Rio Grande ecosystem and informing effective management to recover the Rio Grande Silvery Minnow. The integration of multiple river sciences contributed to valuable insights into the complex dynamics influencing habitat conditions needed by this imperiled species. This was the third Process-Linkage Report produced for this project, which included an assessment of process-linkages for the Angostura Reach of the Middle Rio Grande (Bernalillo, NM to Isleta Diversion Dam). Results from this reach were compared to the Isleta and San Acacia Reaches (Process-Linkage Reports I–II). Future studies are recommended to improve characterization of flow-habitat relationships, assess channel evolution trajectories, evaluate flow management and habitat restoration practices, and further integration across spatial scales (i.e., reach-scale to Middle Rio Grande). Continued progress on the recovery of the Rio Grande Silvery Minnow will depend on the ongoing support of river managers to pursue research and monitoring efforts that inform management of flows, aquatic habitats, and ecological resources.

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KEY TERMS AND DEFINITIONS

This Process-Linkage Report includes terminology used in the disciplines of biology, ecology, engineering, hydrology, and geomorphology. Specific disciplines tend to develop their own perspectives, assumptions, definitions, lexicons, and methods, which can pose a challenge to integrating research efforts (Thoms and Parsons, 2002; Krueger et al., 2016). Accordingly, the terminology and concepts used during interdisciplinary studies should be clearly defined to convey their intended meaning. The following list defines terms and concepts pertaining to the processes and features in the Middle Rio Grande ecosystem that emerged during this interdisciplinary study. This list is not meant to be exhaustive but rather focuses on key terms and those that have potential to cause confusion.

Abundance the number or amount of a species of fish in a particular area. Abundance is estimated using field density measurements (e.g., seine hauls) and expressed as the number of fish per 100 m^2.

Active channel a dynamic geomorphic feature formed by prevailing stream discharges. The active channel is generally narrower than the bankfull channel and defined by a break in bank slope and/or edge of permanent vegetation.

agg/deg lines Approximately equally spaced (~500 ft) transects along the length Middle Rio Grande used to survey channel cross-sections through time. Agg/deg lines were designated by USBR to systematically survey channel cross-sections through time and assess spatiotemporal aggradation and degradation trends in the Middle Rio Grande.

Armoring
- bed: erosion of upper bed sediments, revealing a coarser sediment layer that is resistant to erosion for a given discharge or flow regime.
- bank: increase in the stability of a stream bank by increased sediment size, vegetation and root growth, or modification (e.g., rip-rap).

Bankfull discharge (Q_{BF}) the discharge when the stage (height) of a stream is coincident with the uppermost level of the banks – the water level at channel capacity or bankfull stage (Osterkamp, 2008). Bankfull discharge can vary spatially and temporally.

Bio-habitat conditions characteristics (e.g., timing, magnitude, and duration) of specific aquatic habitat types, such as shallow, low-velocity areas, suspected to determine ecological responses (e.g., fish density) via biophysical interactions with focal species.

Channel aggradation or degradation an increase (aggradation) or decrease (degradation) in the bed elevation of a stream over time.

Channel or stream evolution the morphological response of channel geometry and planform to natural or anthropogenic factors through time.

Channel incision synonymous with channel degradation; decrease in bed elevation over time.

Channelization engineering practices that modify the geometry (width, depth, length) and/or planform of a stream for human purposes (e.g., flood protection, flow conveyance, navigation). Examples of channelization activities include bed and bank armoring, levee construction, and deepening/widening/narrowing/straightening of the channel.

Connectivity
- Lateral: hydrologic connection between the river channel and floodplain that facilitates the movement of fish between these areas.
- Longitudinal: hydrologic connection between upstream and downstream reaches of a river that facilitates the movement of fish between these areas.
- Vertical: hydrologic connection between surface water and groundwater.

Conveyance a measure of the amount of water that can pass through a channel cross-section without inundating higher surfaces (i.e., flooding; Osterkamp, 2008).

Critical habitat the specific geographic area(s) that contain features essential to the conservation of an endangered or threatened species that may require special management and protection. Critical habitat is a term defined by the U.S. Endangered Species Act; designations are made by the U.S. Fish and Wildlife Service. For the Middle Rio Grande, critical habitat defines the length of river and lateral extent (width); the lateral extent includes areas bounded by existing levees or the 300 ft of riparian zone adjacent to each side of the bankfull stage of the river.

Depletion a regulatory policy term used by the New Mexico Office of the State Engineer to estimate the approximate volume of water lost to evaporation during conveyance or storage surface water resources in areas greater than 600 feet from the centerline of the Rio Grande.

Ecological Relationship interaction between an ecological variable (e.g., fish density) and an environmental parameter (e.g., flow or habitat metric).

Ecosystem the complex of biotic populations, the biophysical (environmental) constraints on the biotic populations, and the ability of the complex to function as an ecological unit within a specified area or part of a watershed (Osterkamp, 2008).

Estimated density E(x) measure of fish abundance that accounts for measurement biases (e.g., zero inflated data) using appropriate statistical modeling techniques (e.g., mixture models). Generally expressed as the number of fish per 100 m^2.

Exceedance probability the probability, or likelihood, that the peak discharge of a designated flood event will exceed a specified discharge within some standard period of time, generally a year (Osterkamp, 2008).

Flood relatively high streamflow that overtops the natural or artificial banks in any reach of stream; any flow that inundates the floodplain (Osterkamp, 2008).

Floodplain land adjacent to a stream channel that is inundated at discharges greater than bankfull (Q_{BF}).

Flow duration the percentage of time that a specified discharge is equaled or exceeded.

Flow metric a value derived from stream gaging records that is representative of specific streamflow conditions (e.g., timing, magnitude, and duration) and can be used to evaluate relationships between streamflow conditions and an ecological variable (e.g., fish density or occurrence).

Flow regime the pattern of streamflow over time, generally described in terms of the magnitude, frequency, duration, timing, and rate of change of hydrologic events (e.g., peak and low flows) for a given location or length of stream.

Fossilized channel a stream planform that does not experience considerable lateral movement through time. Distinct from channelization, however, a fossilized channel can form within a channelized reach.

Fragmentation the physical division of a river into discrete reaches by instream barriers (e.g., dams, diversion structures, and culverts). Fragmentation reduces longitudinal connectivity.

Habitat (aquatic) the aquatic environments where an organism completes necessary aspects of its life history (e.g., spawning, feeding/rearing).

Habitat availability herein refers to the normalized stream areas (i.e., area per length) meeting hydraulic criteria (i.e., water velocity and depth) specified as physically suitable for Rio Grande Silvery Minnow. Relationships between discharge and habitat availability (i.e., flow-habitat curves) were derived via hydraulic modeling methods.

Habitat conditions the physical and biological characteristics of aquatic habitats. The habitat conditions required by an organism can vary by life-stage (e.g., larvae, juvenile, and adult).

Habitat complexity (or heterogeneity) a measure of the diversity of habitat types or characteristics within a given spatial unit.

Habitat suitability (habitat criteria) a measure of the adequacy of habitat conditions (physical and/or biological) to meet the ecological needs of a given life-stage of an organism.

Life history the pattern of an organism's survival through its life-stages (i.e., reproduction through adulthood, senescence, and death).

Life-stages the distinct phases of an organism's growth and development. For Rio Grande Silvery Minnow, principal life-stages herein are egg, larva, juvenile, and adult.

Mass curve the cumulative sediment discharge, expressed as mass over time. The slope of the mass curve represents the average sediment transport rate (mass per time) during the specified period.

Double mass curve the cumulative sediment discharge versus cumulative stream discharge. The slope of the double mass curve represents the mean sediment concentration during the specified period.

Mesohabitat a discrete unit of habitat that contains similar physical characteristics (e.g., velocity, depth, and substrate). Mesohabitat types monitored in the Middle Rio Grande include: runs, pools, backwaters, and shoreline associations (e.g., shoreline pool).

Morpho-dynamics the linked hydrologic and fluvial geomorphic processes that determine channel and floodplain morphology through space and time. Morpho-dynamics occur across multiple spatial and temporal scales. Synonymous with hydro(geo)morphology, ecogeomorphology, and other interdisciplinary terms used to describe the suite of hydrologic and geomorphologic processes that occur within catchments and their river systems (Gurnell et al., 2016).

Morphology (fish) describes the form and structure of fish.

Morphology (fluvial) describes the form and structure of a stream or river channel.

Perched channel a condition that occurs when aggradation of the main channel over time causes the elevation of the bed and banks to become higher than the surrounding floodplain.

Population dynamics the patterns of population structure (e.g., size and age composition) for a given species or assemblage that occur through time.

Probability of occurrence δ the probability of a fish species occurring at a particular location.

Processes the movement of or changes to parts and features of the river system, typically measured as rates (Beechie et al., 2010). Examples include sediment transport, channel evolution, floodplain inundation, surface-groundwater interactions, and riparian colonization.

Process-linkages (or linkages) the mechanisms by which morpho-dynamics (i.e., hydrologic-geomorphic processes) and ecological processes affect one another.

Reach a spatial unit of stream length. For this study, reach refers to stream lengths bounded by diversion dams and/or study area boundaries.

Recruitment the survival of fish to adulthood. Recruitment may also be specified for a given life stage (e.g., larval or juvenile recruitment), implying survival to that life-stage but not necessarily to adulthood.

Recurrence interval the average interval of time, generally expressed in years, within which, for example, the magnitude or discharge, of a given flood will be equaled or exceeded (Osterkamp, 2008).

Riparian colonization and succession the process of change in the structure and composition of riparian vegetation over time. Includes encroachment of vegetation into the floodplain and active channel.

Sediment load the mass of sediment passing a channel cross-section over time. Typically approximated as the product of suspended sediment concentration and discharge. Sediment load is generally described in terms of two components, the bed load (transported along the streambed; coarse grained sediments) and suspended load (transported in suspension; fine-grained sediments).

Sediment regime the pattern of sediment inputs, storage, and transport for a specified location or spatial unit through time.

Sediment transport the processes by which sediment is eroded, moved, or deposited along a stream channel and its floodplain by hydrodynamic and gravitational forces.

Seining (seine haul) standardized method for surveying fish species composition in the Middle Rio Grande. A small mesh seine (net strung between two poles) is rapidly drawn through discrete mesohabitats; fish collected in the seine are identified to species, enumerated, and recorded. The length and width of the seine haul are used to quantify sampling effort (i.e., m^2 seined per sampling period).

Spring runoff In the Middle Rio Grande, elevated and prolonged streamflow that typically occurs in the spring (ca. April–June) corresponding to snowmelt runoff in the headwaters.

Subreach a spatial unit of stream length; finer scale than reach. For this study, subreaches are delineated by distinct changes in geomorphic characteristics (e.g., width, slope, confluences) or infrastructure locations (e.g., bridges).

Time Integrated Habitat Metric (TIHM) a metric developed to represent the timing and persistence of hydraulically suitable habitats corresponding to life-stages of Rio Grande Silvery Minnow (this study) – Integral of habitat availability over time for a specified period.

APPENDIX A
SUPPLEMENTARY RESULTS

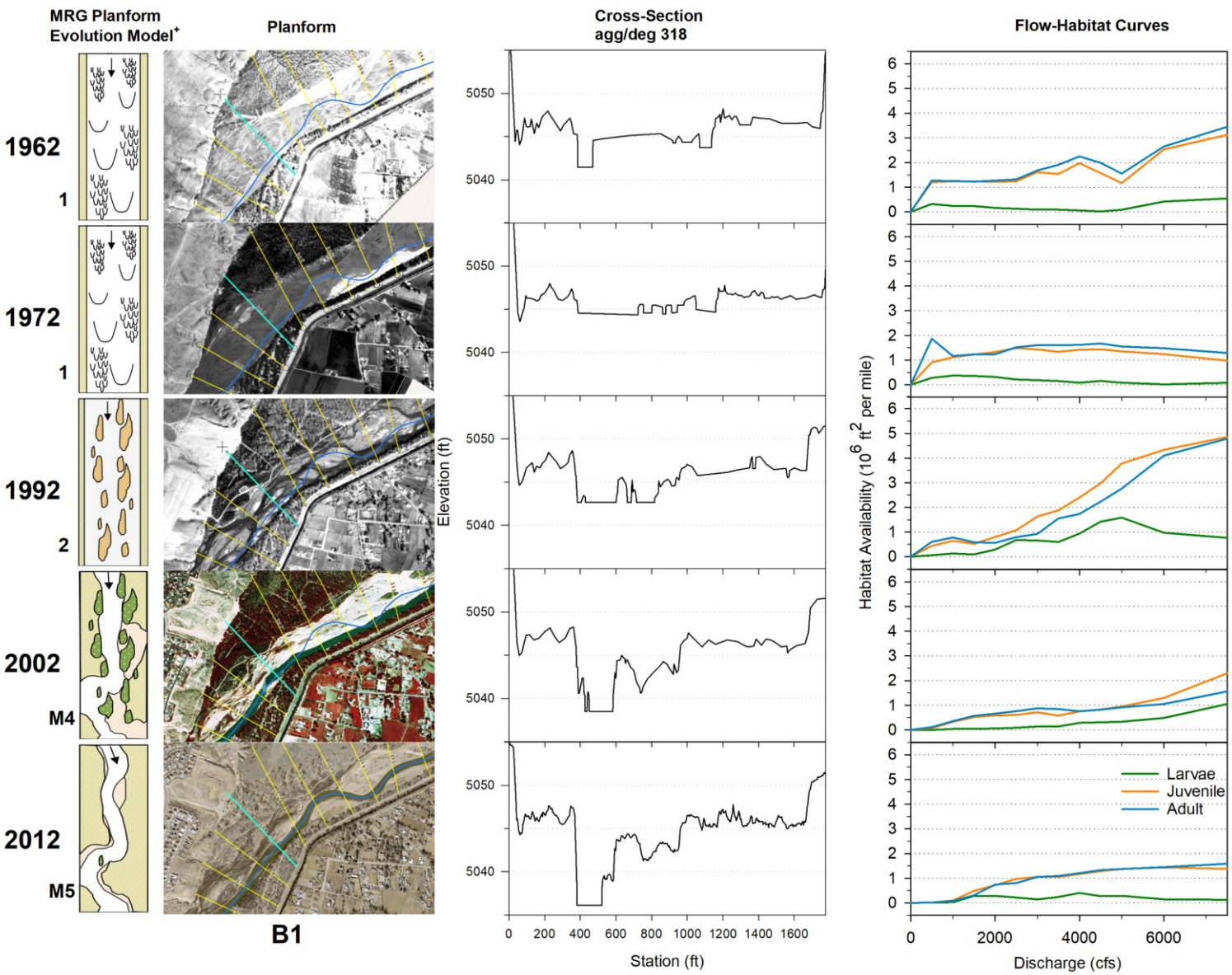

Figure A-1. Channel-habitat evolution model for subreach B1. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

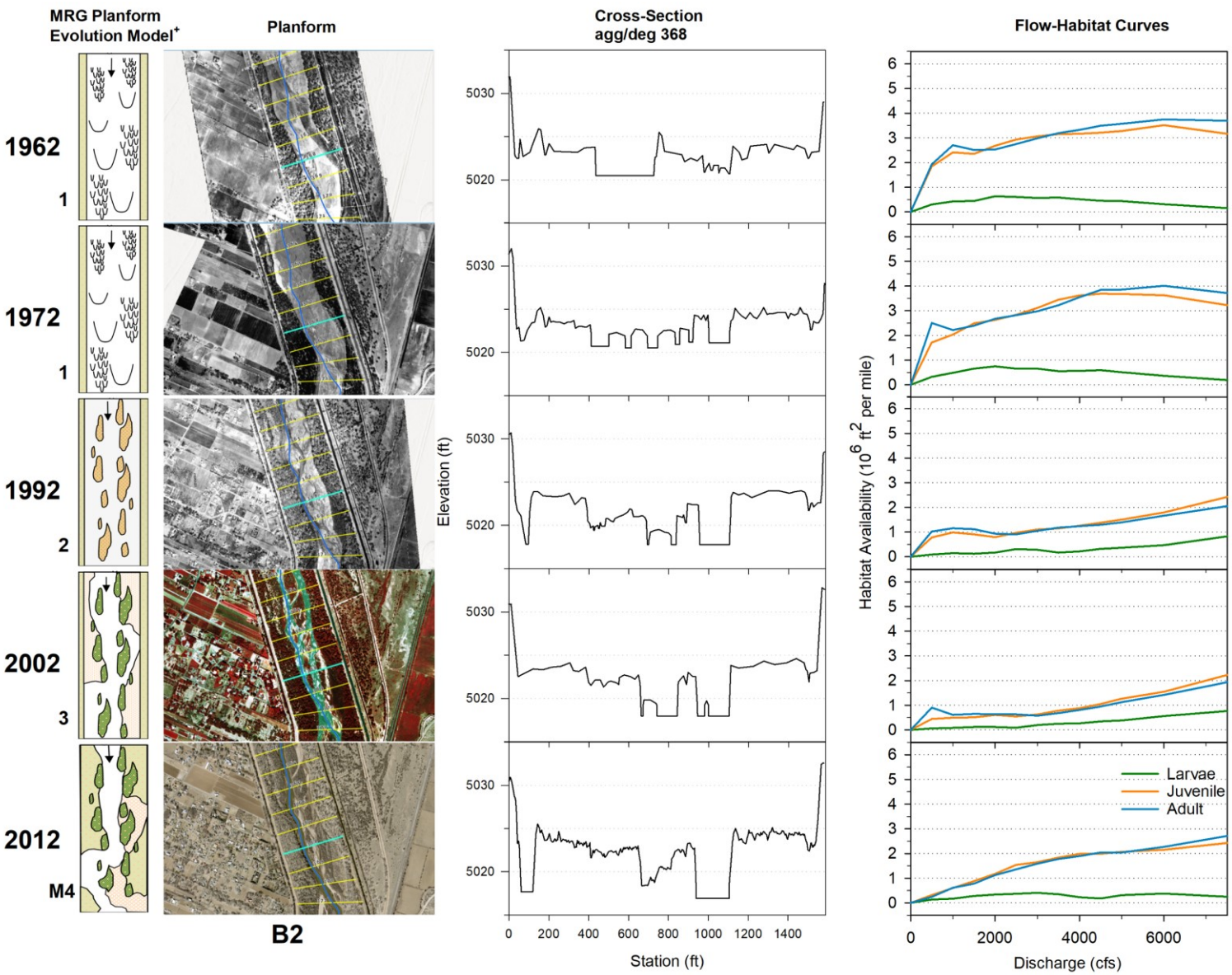

Figure A-2. Channel-habitat evolution model for subreach B2. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

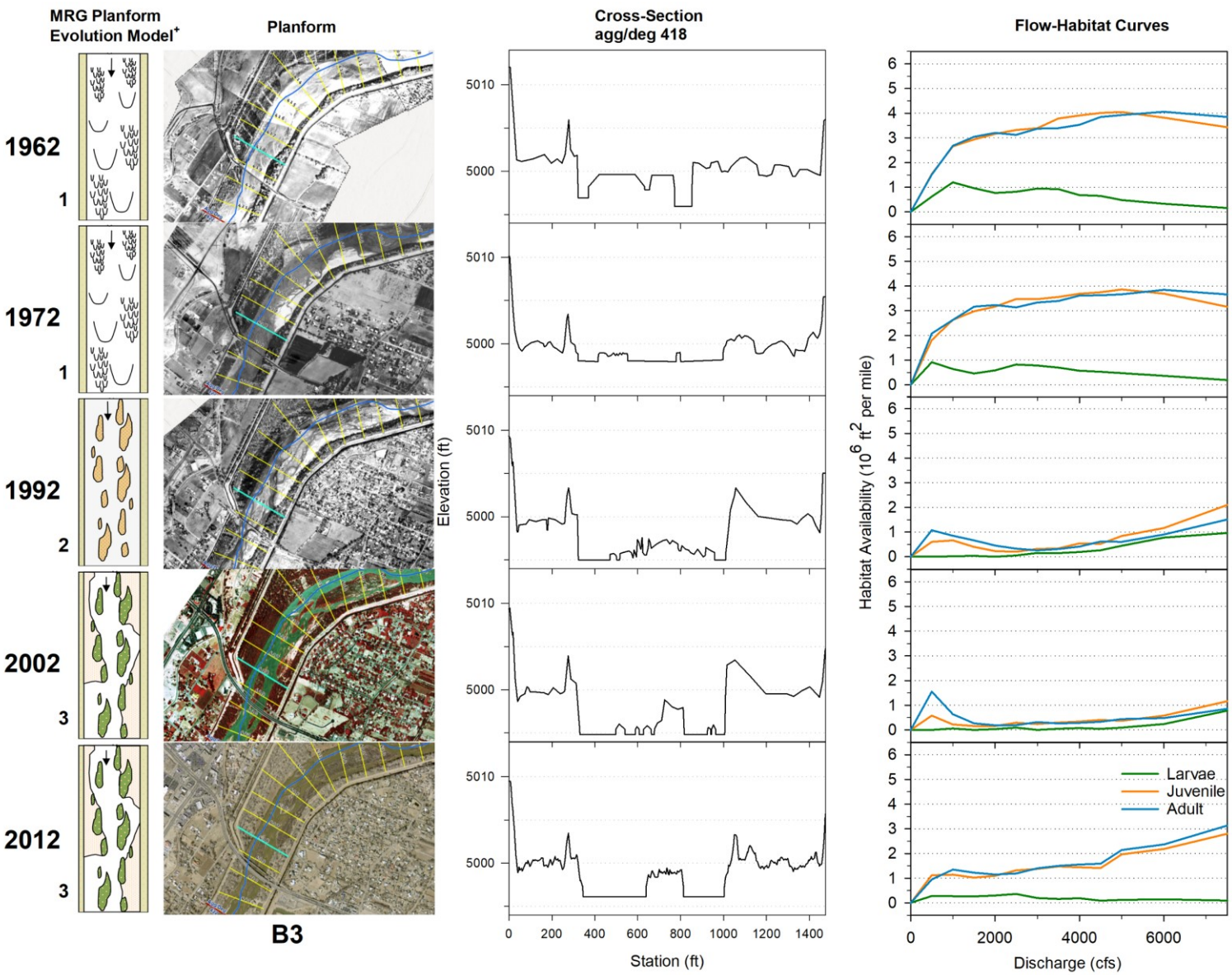

Figure A-3. Channel-habitat evolution model for subreach B3. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

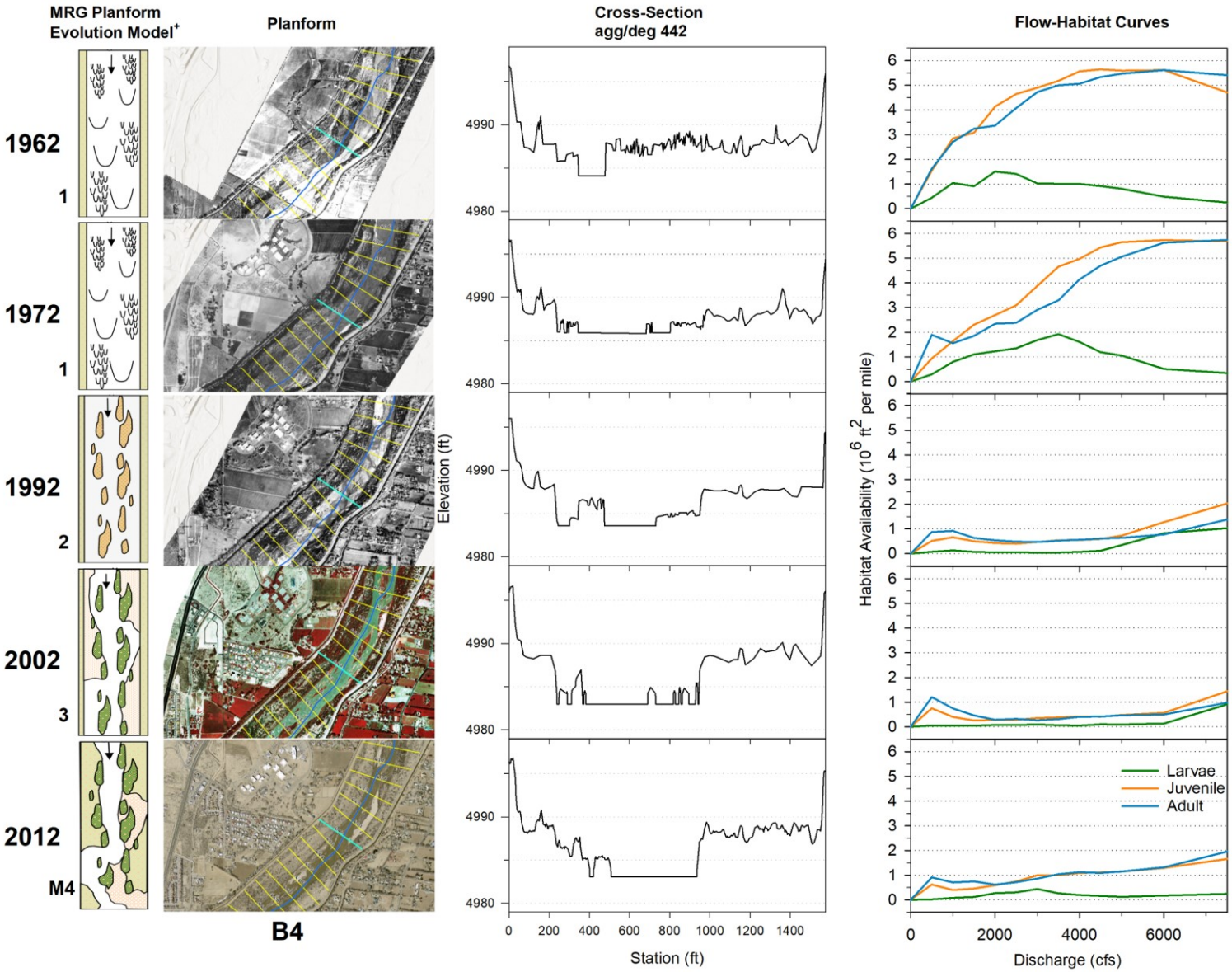

Figure A-4. Channel-habitat evolution model for subreach B4. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

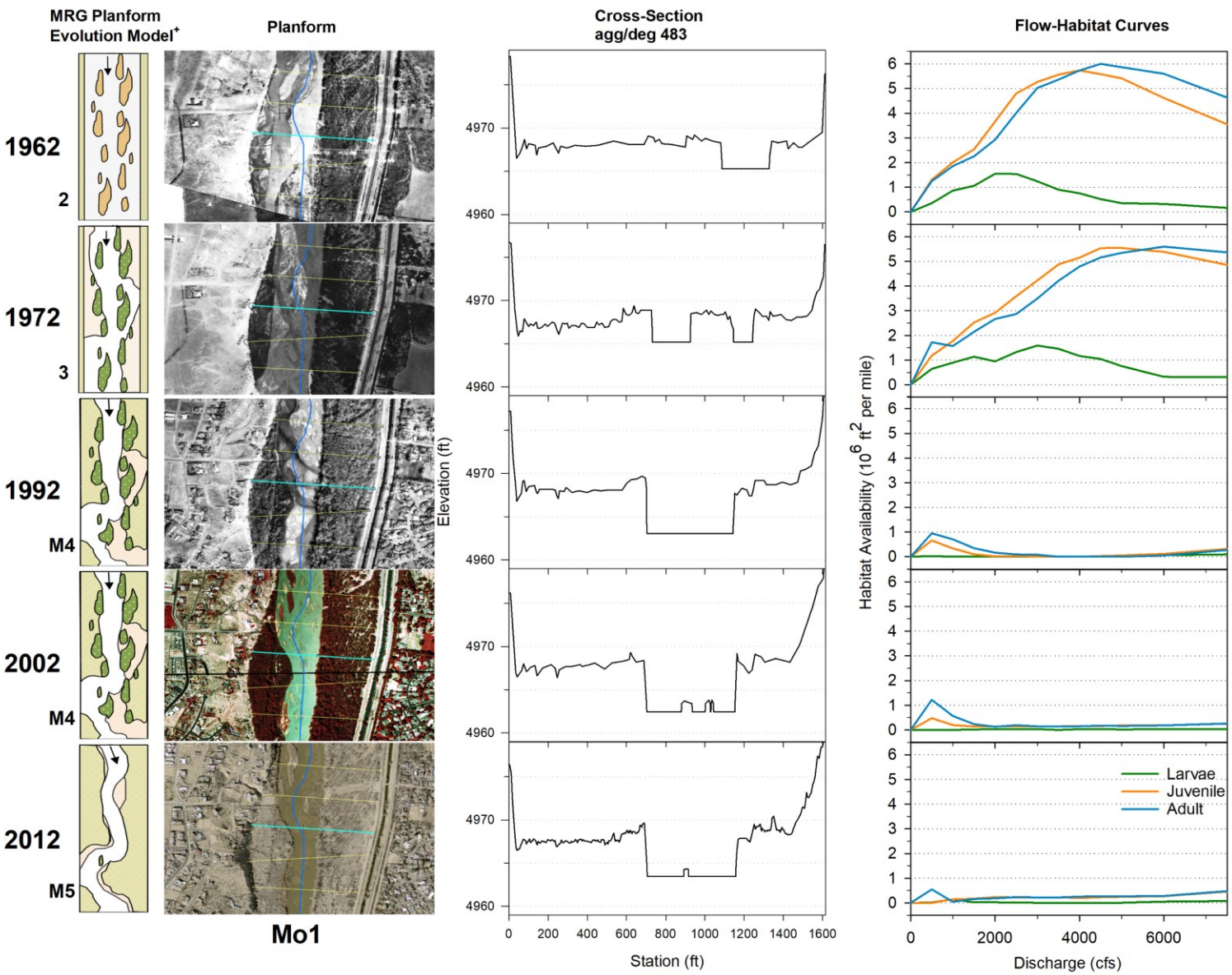

Figure A-5. Channel-habitat evolution model for subreach Mo1. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

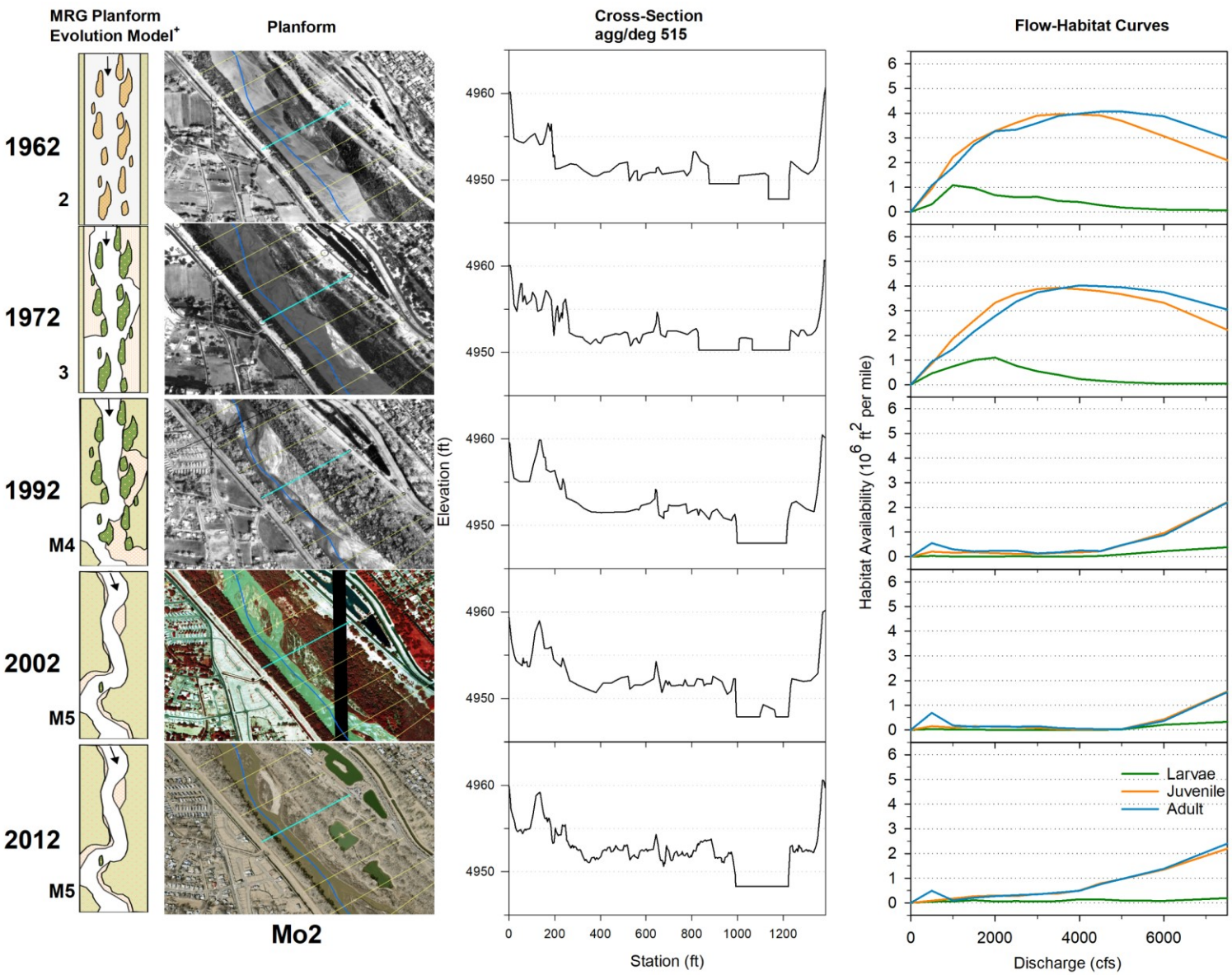

Figure A-6. Channel-habitat evolution model for subreach Mo2. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

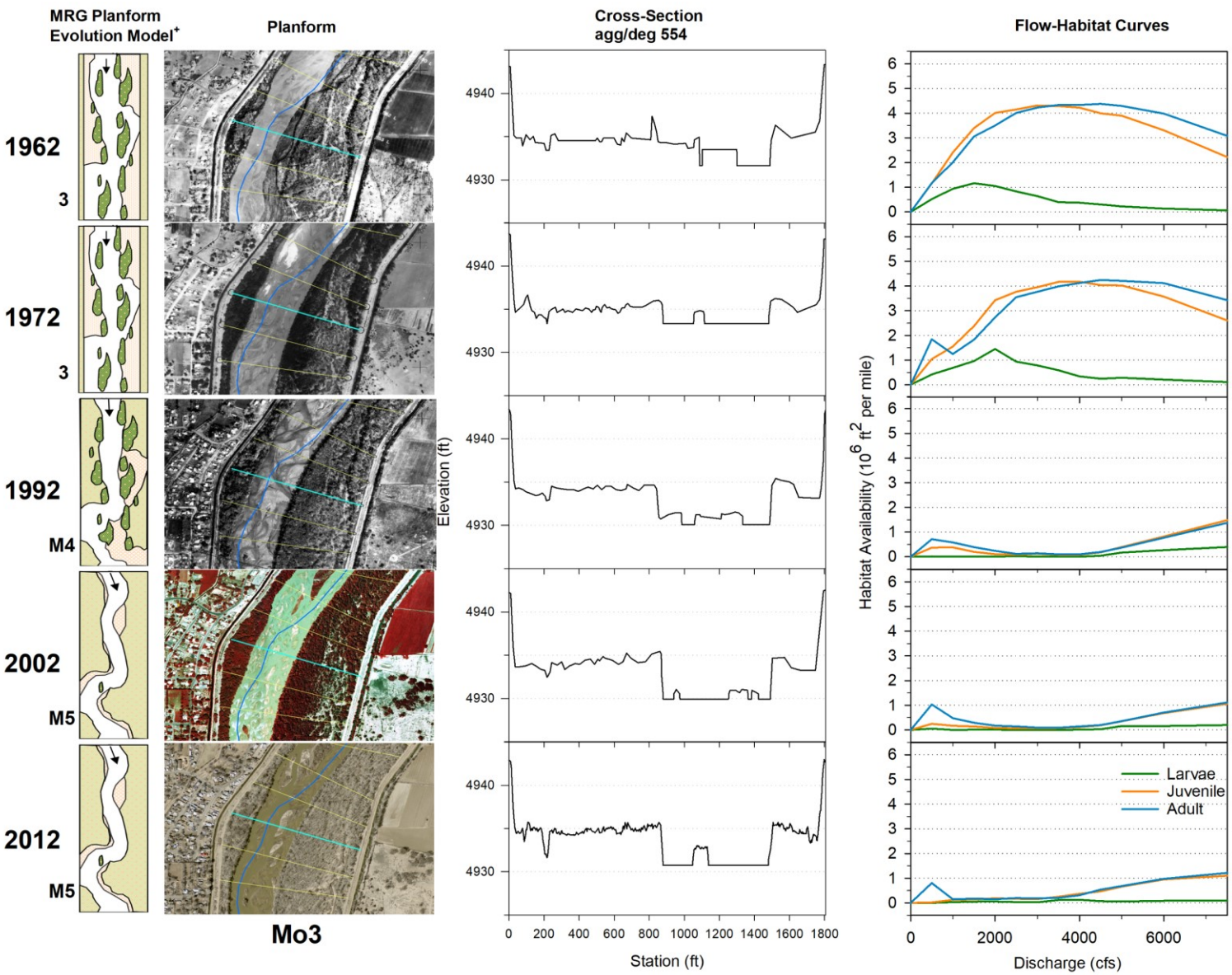

Figure A-7. Channel-habitat evolution model for subreach Mo3. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

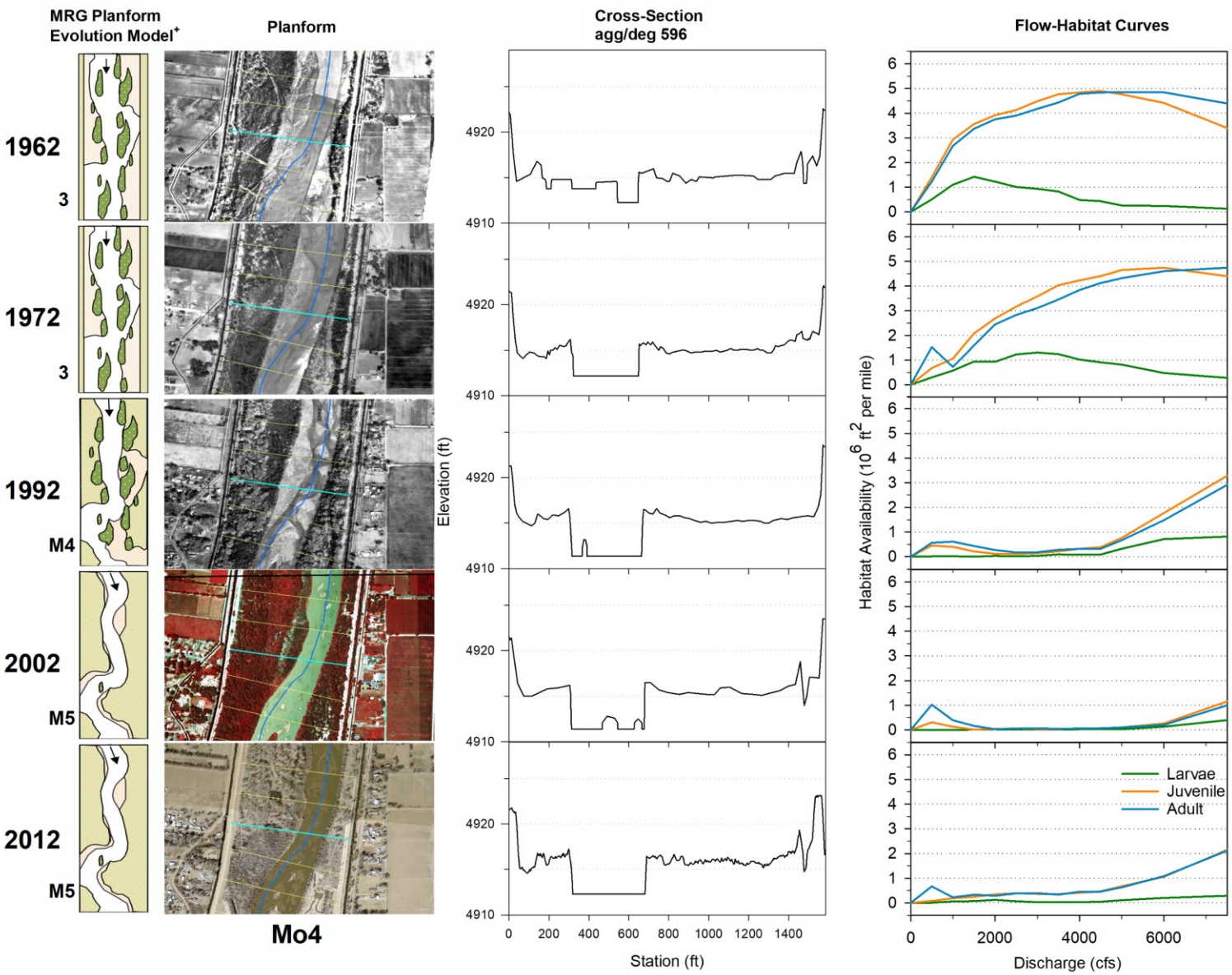

Figure A-8. Channel-habitat evolution model for subreach Mo4. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

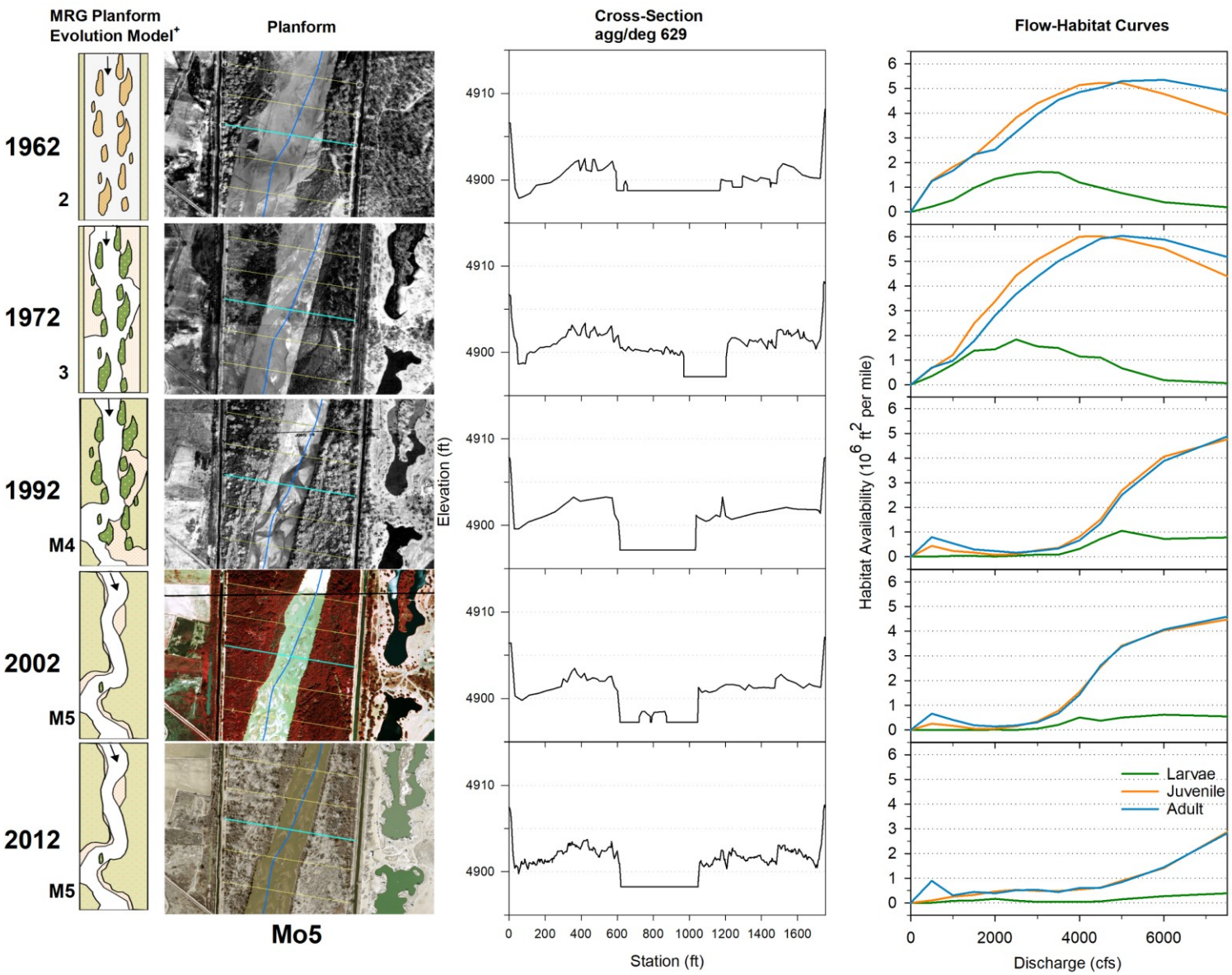

Figure A-9. Channel-habitat evolution model for subreach Mo5. Stages of the planform evolution model (Massong et al., 2010) were estimated from geometry and planform data. Flow-habitat curves corresponding to each stage are shown at right.

APPENDIX B
RIO GRANDE SILVERY MINNOW POPULATION MONITORING SUMMARY

RIO GRANDE SILVERY MINNOW POPULATION MONITORING SUMMARY

This summary describes the Rio Grande Silvery Minnow Population Monitoring Program as included in the Rio Grande Silvery Minnow Biology and Habitat Syntheses (Mortensen et al., 2019). Efforts are ongoing, yet for reporting purposes, the study is briefly summarized for the specified period (1993–2017). However, for more detailed descriptions of study design and specific modifications, sampling and data analysis methods, and Rio Grande Silvery Minnow Population Monitoring results, refer to annual reports submitted to USBR (Albuquerque Area Office) by American Southwest Ichthyological Researchers (ASIR; e.g., Dudley et al., 2020).

Rio Grande Silvery Minnow Population Monitoring Program Overview

The Rio Grande Silvery Minnow Population Monitoring Program is an ongoing long-term systematic monitoring study of the Middle Rio Grande fish community conducted since 1993. This effort provides an annual assessment of recruitment of the Rio Grande Silvery Minnow, a basis for comparing changes in recruitment among years, and timely information on the species conservation status that is especially vital during periods of reduced abundance and occurrence. Original site locations (1993) were based on spatial distribution, site accessibility, relative permanence of flow, presence of reasonably diverse instream habitat (i.e., no highly channelized sites), and resource agency needs.

Since initiation of the Rio Grande Silvery Minnow Population Monitoring Program, there have been numerous changes in both the composition of Middle Rio Grande stakeholders as well as the information needs of resource agencies. This study was designed for the purpose of monitoring long-term trends of the Middle Rio Grande fish community for USBR and the New Mexico Department of Game and Fish. Since then, aspects of the Rio Grande Silvery Minnow Population Monitoring Program have been modified to meet resource agency needs for this endangered species and address diverse aspects of monitoring methodology and statistical analyses (Hubert et al., 2016; Dudley et al., 2018).

Several key components, specific to the Rio Grande Silvery Minnow, that have been added to this study include: (1) evaluating the influence of discharge patterns on population fluctuations, (2) determining general mesohabitat use patterns, (3) documenting changes in relative abundance among fish species, (4) determining variation in density estimates based on repeated sampling, and (5) evaluating changes in site occupancy status across years. The Rio Grande Silvery Minnow Population Monitoring Program has maintained continuity between past and ongoing sampling efforts while incorporating methodological modifications, which has resulted in a rigorous long-term dataset.

Sampling Design and Modification

The Rio Grande Silvery Minnow is a short-lived species and large-scale fluctuations in the abundance and composition of age-classes can occur in only a few months. Sampling frequency targets seasonal and annual variation in the abundance and occurrence of the Rio Grande Silvery Minnow. While data collection has occurred annually since 1993, permitting issues precluded sampling in 1998 and funding issues resulted in reduced sampling during 2009 (Table B-1).

Monthly sampling efforts (April–October) target recruitment and survival of young-of-the-year individuals in relation to often unpredictable and dynamic environmental conditions (e.g., spring runoff, monsoons, irrigation withdrawal) that occur during this period. October sampling data have been collected consistently since 1993 to assess inter-annual population trends. Fish present in October have survived the cumulative effects of the preceding environmental conditions (e.g., spring runoff, monsoons, river drying) and constitute the reproductive cohort heading into the following spring. Further, conditions during October (e.g., streamflow, water temperature, and turbidity) are quite stable and suitable for efficient sampling, as compared to other times of the year (e.g., spring runoff or summer monsoons), making it the most informative month for evaluating long-term population trends of the Rio Grande Silvery Minnow. Monitoring sites are also sampled repeatedly, over four consecutive days, in November (‘repeated’ sampling; 2005–2017), to characterize sampling variation, estimate site occupancy rates, and assess site-specific colonization and extinction trends.

Table B-1. The Rio Grande Silvery Minnow Population Monitoring Program data summary 1993–2017. Sampling frequency indicates the number of sampling events per year. Sampling distribution indicates the number of sites per reach. Monitoring did not occur in 1998. 'Repeated' sampling began in 2005. 'Additional' sampling be

Year	Month												Sampling	Sampling Distribution			
—	J	F	M	A	M	J	J	A	S	O*	N	D	Frequency	Angostura	Isleta	San Acacia	MRG
1993		■			■		■			■			4	5	4	7	16
1994		■			■		■			■			4	5	4	7	16
1995		■			■		■			■			4	5	4	7	16
1996		■			■		■			■			4	5	4	7	16
1997		■					■	■		■			4	5	2	8	15
1998													—	—	—	—	—
1999		■			■		■	■		■		■	6	5	2	8	15
2000		■		■		■		■		■		■	6	5	2	8	15
2001		■		■		■		■		■		■	6	5	6	9	20
2002	■	■	■	■	■	■	■	■	■	■	■	■	12	5	6	9	20
2003	■	■	■	■	■	■	■	■	■	■	■	■	12	5	6	9	20
2004	■	■	■	■	■	■	■	■	■	■	■	■	12	5	6	9	20
2005	■	■	■	■	■	■	■	■	■	■	R	■	12	5	6	9	20
2006	■	■	■	■	■	■	■	■	■	■	R	■	12	5	6	9	20
2007		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2008		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2009									■	■	R	■	4	5	6	9	20
2010		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2011		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2012		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2013		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2014		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2015		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2016		■		■	■	■	■	■	■	■	R	■	10	5	6	9	20
2017				■	■	■	■	■	■	■	R		9	5(5)	6(4)	9(1)	20(10)

Legend

■	Denotes sampling event within specified year and month
*	October sampling data used to assess annual population trends
R	Repeated sampling conducted during November to estimate site occupancy rates and characterize sampling variation
(#)	Denotes the number of 'additional' sampling sites by reach – additional sites sampled twice annually (April and October)

Sampling of the Middle Rio Grande fish community has occurred systematically at 15–30 sites between Angostura Diversion Dam and Elephant Butte Reservoir since 1993. The Cochiti Reach, Cochiti Dam outfall to Angostura Diversion Dam, is not currently sampled because of limited access; the last comprehensive survey of this reach by ASIR personnel was in 1994 (Platania 1993b, 1995b). Since 2001, 20 'standard' sampling sites have been monitored in the Angostura (n=5), Isleta (n=6), and San Acacia (n=9) reaches of the Middle Rio Grande (Table B-1; Figure B-1).

While most sampling sites have been consistently monitored since 1993, several localities have been added (e.g., to increase spatial coverage within or among reaches) or removed (e.g., loss of continuous access). Between 1993 and 2000, monitoring occurred in the Angostura, Isleta, and San Acacia reaches at 15 or 16 sites compared to 20 sites from 2001 to 2016. In 2017, 10 'additional' sampling sites were incorporated in the study. The additional sampling locations were included to reduce spatial distance between sites and provide 10 sites per reach (regardless of the differential reach lengths). Additional sites are sampled twice annually (April and October) and seamlessly integrate with the standard monitoring efforts. Also in 2017, 'replacement' sites were incorporated in the study design to accommodate periods of river drying. This new protocol requires that a wetted replacement site be sampled for each dry site that is encountered within each reach. While the recent modifications (2017) were meant to provide additional data to address concerns regarding the spatial distribution of sampling sites and the importance of river drying events, it is still too early to evaluate the utility of these modifications.

Methods

Sampling methods have remained consistent throughout the duration of the Rio Grande Silvery Minnow Population Monitoring Program (1993–2017). Fish are collected by seining, an efficient and well-established sampling method in sand-bottomed rivers such as the Rio Grande where habitat complexity is relatively low (Rabeni et al., 2009). A small-mesh seine (3.1 m x 1.8 m; ca. 4.8 mm mesh) is used to collect small-bodied fish (i.e., juveniles and adults <120 mm TL) and a fine-mesh seine (1.2 m x 1.2 m; ca. 1.6 mm mesh) is used to collect larval fish. Each seine haul constitutes an individual sample, and 20 samples (18 small-bodied, 2 larval) are taken at each site (20 samples x 20 sites =400 samples/month). Small-bodied fish are identified and enumerated by sample (1–20) and those results are recorded in the field. Additionally, the mesohabitat type and seine haul length (<15 m) are also recorded for each sample. All Rio Grande Silvery Minnows are measured, identified to age-class (based on reach-specific age-length relationships and date of collection), and examined for Visible Implant Elastomer (VIE) tags indicative of hatchery-reared fish. All sampled fish are temporarily held in a live-well at the site and released unharmed at the conclusion of sampling. Fish too small to be accurately identified in the field (e.g., larvae or early juveniles) are preserved in 10% formalin and subsequently processed in the laboratory (Division of Fishes, Museum of Southwestern Biology, University of New Mexico) by personnel specifically trained to identify larval fishes of the Middle Rio Grande. Digital photographs and selected water quality parameters are also recorded at each site.

Sampling data are normalized to density for statistical analyses. Density (i.e., catch-per-unit-effort [CPUE]) is computed by dividing the number of individuals captured by the area sampled, multiplied by 100 (i.e., CPUE = fish per 100 m^2). Area sampled (i.e., effort; m^2) is calculated by multiplying seine haul length by the respective sampling width (i.e., small-mesh seine [2.5 m] and fine-mesh seine [1.0 m]). Sampling effort for this study is substantial (400 seine hauls ≈10,000 m^2 sampled/month). As different sampling equipment and protocols are used to capture specific developmental stages, larval and small-bodied fish densities are analyzed independently. Individuals marked with VIE tags (i.e., hatchery-reared fish) are excluded from analyses of long-term population or occupancy trends.

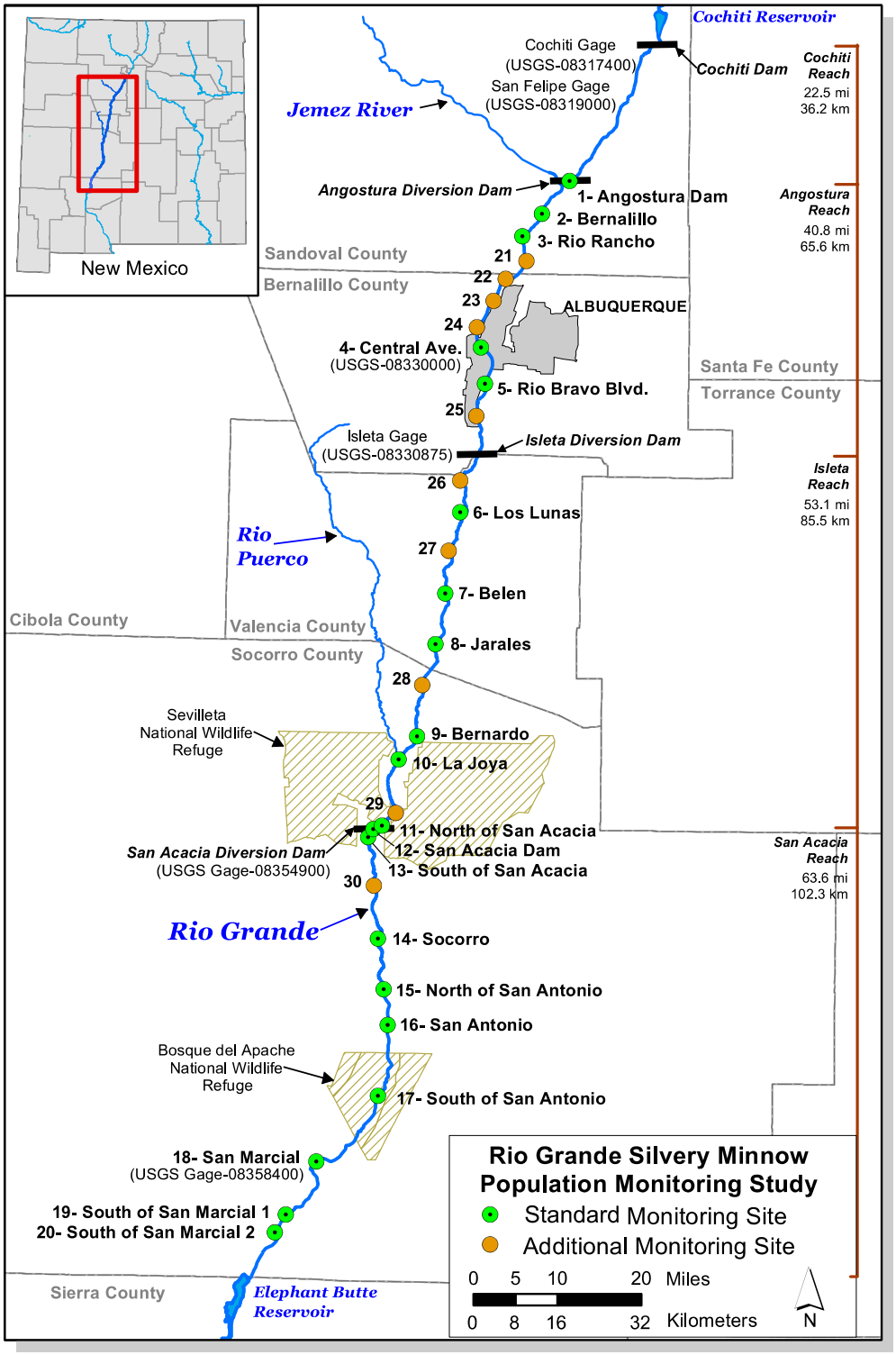

Figure B-1. Map of the study area for the Rio Grande Silvery Minnow Population Monitoring Program (from Dudley et al., 2020). Currently, 'standard' sites are sampled monthly April–November and 'additional' sites are sampled twice annually (April and October).

Rio Grande Silvery Minnow Population Trends (1993–2017)

Temporal Trends

Over the past two decades, there have been large inter-annual fluctuations in the estimated densities of the Rio Grande Silvery Minnow (i.e., more than three orders of magnitude [>100,000% increase or >99.9% decrease]). Between 1993 and 1997, estimated densities of the Rio Grande Silvery Minnow in October were relatively high (>10 fish/100 m^2) with the exception of a decrease during 1996. Between 1999 and 2003, estimated densities of the Rio Grande Silvery Minnow declined precipitously and, in 2003, the abundance and occurrence of the Rio Grande Silvery Minnow was too low (i.e., one individual collected from one site) to statistically estimate its density. Densities of this species increased notably during 2004 and 2005 with the latter year producing the highest estimated densities during the tenure of the Rio Grande Silvery Minnow Population Monitoring Program. Between 2006 and 2011, estimated densities fluctuated by an order of magnitude (ca. 1–10 fish/100 m^2). The Rio Grande Silvery Minnow was not collected at any of the sampling sites in 2012 or 2014. Between 2015 and 2017, estimated density increased dramatically and, in 2017, it was among the highest values observed during the monitoring period (e.g., 1993–1995, 1997, 2005, 2007–2009). Over the duration of the study (1993–2017), there have been wide and frequent fluctuations of the population.

Seasonal trends are apparent for different developmental phases and age-classes of the Rio Grande Silvery Minnow. Densities of larval individuals increase following spring spawning, reaching their highest levels in June and July, but tend to drop precipitously by July and August. Declines in the densities of larval fish can be attributed to: 1) progression through the larval developmental phase (i.e., larvae to juvenile), and 2) high mortality during the larval phase. Age-0 fish are typically at relatively low densities in June, reach their highest densities in July and August, and decline during September and October. Comparison of October and November sampling efforts revealed similar trends in the estimated densities of the Rio Grande Silvery Minnow over time (2005–2017), however, estimated densities tended to be somewhat higher in November. This pattern may be explained by the tendency of the Rio Grande Silvery Minnow to aggregate more often in deeper and lower velocity habitats during winter when water temperatures are lower (Dudley and Platania 1997) than during months when water temperatures are warmer. Age-1+ fish are relatively rare throughout the year and across the study period. From February to May, Age-1+ fish compose the entire population. Following years with adequate spring spawning flows, newly spawned individuals (i.e., Age-0 fish) compose the vast majority of the population from June to November. Seasonal trends in the abundance of age-classes, documented by the Rio Grande Silvery Minnow Population Monitoring Program, support seasonal patterns of age-class structure.

Spatial Trends

Densities of the Rio Grande Silvery Minnow are spatially variable across sites and reaches within the Middle Rio Grande. Sampling efforts during October (1993–2017) indicated that the highest densities of this species were nearly always in the Isleta or San Acacia reaches. This longitudinal pattern has persisted even though upstream reaches have been regularly augmented with large numbers of hatchery-reared fish since 2001 (Archdeacon, 2016). Exceptions to this pattern occurred in years when flows in the San Acacia Reach were unusually low during spring and summer (e.g., 2002–2003 and 2012–2013). The general pattern of increasing densities downstream is likely explained by the cumulative downstream transport of their propagules (i.e., eggs and larvae) past instream barriers (Dudley and Platania, 2007). Additionally, river channelization, habitat degradation, reduced floodplain connectivity, and reductions in suspended sediments downstream of Cochiti Dam likely limit the availability of suitable habitat for the successful retention and recruitment of early life stages, especially in the upstream Cochiti and Angostura reaches (Richard and Julien, 2003; Massong et al., 2006). These factors likely influence spatial trends of larval Rio Grande Silvery Minnows.

Based on recent data (2016–2017), there appear to be differences in the distribution and abundance patterns for larval and juvenile/subadult Rio Grande Silvery Minnows. Although larval fish densities were similarly elevated in all surveyed reaches following spring spawning, densities of juvenile/subadult Rio Grande Silvery Minnows were consistently lower in the Angostura Reach throughout summer and autumn. Densities of juvenile/subadult fish also peaked somewhat later, and subsequently remained higher, in the San Acacia Reach than in the two upstream reaches.

These findings suggest that: (1) survival rates for young were relatively lower in the Angostura Reach than in the two downstream reaches, (2) young were progressively dispersing downstream, either passively or actively, during the spring and summer, or (3) these patterns were caused by some combination of the first two factors (Dudley et al., 2018). These seasonal reach-specific patterns are based only on recent data, however, as delineating early life-stages (e.g., separating late stage mesolarvae from early-stage juveniles) has only been funded since 2016. Also, these potential patterns do not account for variation across sites within a reach (i.e., no confidence intervals), and could change depending on annual spring and summer flow conditions. Further, densities of the Rio Grande Silvery Minnow across reaches are not a direct reflection of population size, as the amount of wetted area is often notably higher in the Angostura Reach than in the downstream reaches, which can lead to higher population estimates despite lower density estimates (Dudley et al., 2012). Thus, possible differences in recent reach-specific densities of the Rio Grande Silvery Minnow should be interpreted cautiously.

Relationships Between Population Trends and Hydrologic Conditions

Ecological and statistical models are used to quantitatively assess the effects of various environmental variables on long-term trends in the abundance and occurrence of the Rio Grande Silvery Minnow. Robust modeling approaches are required to account for the large proportion of zeros, which are especially common in ecological datasets of rare or imperiled species. Mixture models (e.g., combining a binomial distribution with a lognormal distribution) are particularly effective for modeling zero-inflated data (White 1978; Welsh et al., 1996; Fletcher et al., 2005; Martin et al., 2005) and for evaluating the effects of environmental covariates on population parameters. These models are used to estimate parameters for each sampling year based on site-specific sampling data (e.g., n=20 standard sites): estimated density $E(x)$, estimated occurrence probability (δ), estimated lognormal density (μ), and standard deviation of the estimated lognormal density (σ). Population parameters provide a basis for identifying and assessing ecological relationships between population dynamics and environmental conditions.

Assessing the influence of environmental variability on the Rio Grande Silvery Minnow population lends insight to important mechanisms that regulate abundance and occurrence. Various hydrologic covariates (e.g., spring and summer flow metrics) have been assessed individually to determine their effectiveness in explaining the variation in estimated population parameters (e.g., density [$E(x)$] and occurrence [δ]) through time. Metrics representing spring runoff conditions (May–June) include maximum discharge and days exceeding threshold discharge values (i.e., >1,000 cfs, >2,000 cfs, and >3,000 cfs). Spring runoff metrics are computed using streamflow data from the Rio Grande at Albuquerque, NM (USGS 08330000). An additional metric representing estimated inundated acreage during peak flows (i.e., mean of the five peak flow days in May; USACE 2010), has been used to assess the influence of spring flooding on long-term population dynamics. Metrics representing low flow conditions during the irrigation season (March–October) include first day with discharge <200 cfs, mean daily discharge, and days below threshold discharge values (i.e., <200 cfs and <100 cfs). Low flow metrics are computed using streamflow data from the Rio Grande at San Marcial, NM (USGS 08358400). These hydrologic covariates are relatively simple and easily obtained metrics, and thus do not entirely capture the temporal (e.g., seasonal) and spatial (e.g., site-specific or reach-specific) heterogeneity of hydrologic conditions that can occur in the Middle Rio Grande. While these limitations are important to recognize, the chosen metrics are crucial for identifying and quantitatively assessing important ecological relationships.

Comparison of hydrologic metrics to changes in density and occurrence of the Rio Grande Silvery Minnow (i.e., $E(x)$ and δ) in October (1993–2017) revealed several strong ecological relationships. Peak discharge and duration of high flows during spawning/rearing season (primarily May–June) were related to increased density and occurrence of this species. In contrast, extended low flows during summer were related to decreased density and occurrence. Modeling these two separate population responses (i.e., density and occurrence) provided valuable insights into long-term population trends for this species. These analyses indicated that elevated and prolonged spring flows were most predictive of range-wide increases in the density and occurrence of the Rio Grande Silvery Minnow over time (Dudley et al., 2018). Similarly, increased numbers of Age-0 Rio Grande Silvery Minnows collected in isolated pools during periodic river drying events from June to October (2009–2015) were closely related to elevated mean May discharge during the same year (Archdeacon, 2016). These assessments identified the impact of seasonal hydrologic conditions on the population dynamics of the Rio Grande Silvery Minnow.

Mesohabitat Use

The Rio Grande Silvery Minnow Population Monitoring Program has also provided qualitative assessment of the mesohabitats most commonly occupied by the Rio Grande Silvery Minnow. While the physical locations of mesohabitats shift considerably over time, especially in a mobile sand-bed river such as the Middle Rio Grande, established sampling protocols for population monitoring ensure that similar mesohabitats (as characterized by water depths and velocities) are consistently sampled across sites and years. Since 2002, a wide variety of mesohabitats has been sampled to provide balanced monitoring for the fish community and all life stages of the Rio Grande Silvery Minnow. Assessment of mesohabitat use over the period of study (2002–2017) has shown notable differences in the estimated densities of the Rio Grande Silvery Minnow among the five different sampled mesohabitats (i.e., backwater, pool, run, shoreline pool, shoreline run). Densities of the Rio Grande Silvery Minnow were typically highest in lower velocity mesohabitats (e.g., backwater and pool) and lowest in higher velocity mesohabitats (e.g., run and shoreline run; Dudley et al., 2018). The general mesohabitat use patterns observed during population monitoring are similar to those documented by past habitat use studies (e.g., Dudley and Platania, 1997).

Analytical Considerations

Analytical considerations discussed herein refer to the analyses performed for both the Rio Grande Silvery Minnow Population Monitoring Program and the Linkage Report (this report).

The mixture models used to estimate densities of the Rio Grande Silvery Minnow in this study employed two separate statistical components, an approach that is particularly effective for modeling zero-inflated ecological data (White, 1978; Welsh et al., 1996; Fletcher et al., 2005; Martin et al., 2005). Logistic regression was used to estimate the annual probability that a site was occupied, and a lognormal model was used to estimate the annual lognormal density based on occupied sites. The two processes (i.e., occurrence [δ] vs. density [μ]) that generated $E(x)$ were clearly separated when using the mixture-model approach (Dudley et al., 2020). Also, it was unnecessary to add some arbitrary positive constant onto observations of zero values, as is commonly done for simple linear regression models using log-transformed data. Further, our approach fully accounts for over-dispersion (e.g., extra-binomial variation around δ, non-constant σ in the lognormal distribution, or additional variation around δ and μ the linear covariate model). Thus, we have produced estimates using a robust, yet highly flexible, approach that avoids many assumptions typically required for traditional statistical analyses (Dudley et al., 2020).

One assumption required for our analyses is that capture probabilities are reasonably similar across sampling sites and years. As mark-recapture or multiple-pass data were not collected as part of this study, this assumption cannot be directly evaluated. However, it seems highly unlikely that pronounced downward density trends were caused by low capture efficiencies, as our methods have remained consistent to ensure that comparable mesohabitats (i.e., depths and velocities) were sampled across different sites and annual flow conditions. As an example, a substantial decline (> 90%) in density between years (e.g., 1995–1996, 2005–2006, and 2017–2018) would require a seemingly unreasonable decrease (> 90%) in capture probability (e.g., 0.5 to 0.01) between those years. Additionally, seining has been shown to be quite effective and reliable in sand-bottomed rivers, such as the Rio Grande, where habitat complexity is relatively low (Rabeni et al., 2009). Thus, it seems more reasonable that any differences in capture efficiencies across sites or years would tend to average out because of the substantial sampling effort required for this study. Further, environmental conditions during October (e.g., water temperatures, flows, depths, velocities, and turbidities) have been quite stable and suitable for efficient sampling as compared to other times of the year (i.e., spring runoff or summer monsoons), making it an ideal time of year for evaluating long-term trends in the occurrence and density of the Rio Grande Silvery Minnow. Finally, we have also maintained a steadfast consistency in our crew leaders, training procedures, and sampling protocols over the past two decades.

Although we used frequentist statistical methods (i.e., mixture models and generalized linear models) to analyze the long-term data in our study, we also evaluated the merits of the Bayesian method of statistical inference. Frequentist and Bayesian approaches both use the same general analytical framework (i.e., parametric likelihood models supplemented with linear covariate models) to generate parameter estimates and make ecological inferences from the data. However, Bayesian techniques rely on subjective assumptions about prior distributions, and require additional Markov chain Monte Carlo

(MCMC) statistical analyses to obtain model estimates (Burnham and Anderson, 2002). Therefore, conducting Bayesian analyses based on a non-hierarchical framework, as was used in our study, will not result in different conclusions, but does raise the issues of including subjective data and interpreting additional statistical results. While the Bayesian approach might seem preferable for reach-specific analyses, using informative priors to substitute for sparse reach-specific data seems contrary to objective monitoring. Thus, we have used the frequentist statistical approach to rigorously analyze long-term trends in the occurrence and density of the Rio Grande Silvery Minnow and evaluate how those trends were affected by environmental changes over time (1993–2019).

Strengths and Limitations of the Rio Grande Silvery Minnow Population Monitoring Program

The sampling design and methodology of the Rio Grande Silvery Minnow Population Monitoring Program has been continually assessed to verify its ability to provide robust estimates of population trends for the Rio Grande Silvery Minnow. The methods used for data analyses are statistically robust (e.g., mixture models) and appropriate for modeling the ecological data collected. Furthermore, recent population monitoring reports have provided numerous comparisons between diverse methods of analysis that support the core methods and results (e.g., estimated density vs. method of moments, sampling-site density data vs. mesohabitat-specific density data, standard sampling vs. repeated sampling, and population monitoring results vs. site occupancy or population estimation results). Also, in 2017, ‘additional’ and ‘replacement’ sampling sites were selected to reduce spatial sampling gaps and address concerns regarding the treatment of sampling data during river drying. This modification produced four different datasets to evaluate sampling design and methods; all four datasets were consistent with the key findings of the long-term study.

Additionally, the Rio Grande Silvery Minnow Population Monitoring Program has addressed issues related to sampling variability. In brief, a negligible proportion of observed temporal variability in the Rio Grande Silvery Minnow density is likely due to sampling variability. By default, natural variability is less than the observed variability. To obtain an unbiased estimate of natural variability, it is necessary to estimate sampling variability. Sampling variability can be estimated by performing multiple sampling events at the same site (i.e., ‘repeated’ sampling). During ‘repeated’ sampling efforts, the 20 ‘standard’ sites are sampled once per day for four days during November. Conducting ‘repeated’ sampling once per year (i.e., November) from 2005 to 2017 provided valuable estimates of the proportion of sampling variation, thereby increasing confidence in the estimated population trends over time. The Rio Grande Silvery Minnow Population Monitoring Program has maintained a strong and defensible basis for assessing seasonal and annual trends in abundance and occurrence of the Rio Grande Silvery Minnow.

While the design and methodology of the Rio Grande Silvery Minnow Population Monitoring Program is statistically rigorous for its intended objectives, limitations arise when monitoring data and results are applied beyond the scope of their intended purpose. For example, using monitoring results (e.g., seine haul densities) to estimate population size violates multiple statistical assumptions and yields inaccurate estimates (Dudley et al. 2012). To provide resource agencies with an accurate and statistically robust estimate of annual population size, as opposed to seasonal and annual trends in abundance and occurrence (i.e., population monitoring), substantial modification to sampling design and methodology was required to develop the Rio Grande Silvery Minnow Population Estimation Program (Dudley et al. 2012). Additional limitations of population monitoring data are related to the high degree of spatial and temporal variability of the data, which is common in ecological applications. The effects of variability can be ameliorated, however, by assessing trends at larger scales (e.g., reach-scale, across months/years), increasing sampling effort (e.g., additional sites/samples), and characterizing variability across sampling occasions (e.g., ‘repeated’ sampling). In particular, the Rio Grande Silvery Minnow Population Monitoring Program has used ‘repeated’ sampling data to evaluate sampling variation and quantitatively assess the level of variance in estimated densities at different temporal and spatial scales (i.e., year, sampling occasion, site, and reach). Results indicate that sampling year accounted for the overwhelming amount of variance and was the most informative factor in explaining changes in the densities of the Rio Grande Silvery Minnow (Dudley et al. 2018). These results suggest that changes in the abundance and occurrence of the Rio Grande Silvery Minnow are much more strongly related to seasonal flow conditions across years, as compared to site-specific or reach-specific conditions. As such, attempting to explain population change by relating site-specific density data to localized conditions or metrics (e.g., site-

specific habitat quality) will almost certainly yield insignificant or fallacious relationships. Therefore, due to the fundamental qualities of these datasets, the sampling protocols used to obtain them, and ecological studies in general, caution should be exercised in any assessment of population monitoring data beyond the given scope of the Rio Grande Silvery Minnow Population Monitoring Program.

Reach-Scale Dataset

Population parameters for the Rio Grande Silvery Minnow, based on October sampling-site data (i.e., estimated density *E(x)*, occurrence probability δ, and lognormal density μ), were calculated for individual reaches (i.e., reach-scale – Angostura, Isleta, and San Acacia reaches) for analytical purposes of this study. These values are provided here for comprehensive reporting (Table B-2, B-3, B-4) – these data should be used cautiously as they are affected by downscaling from range-wide estimates (e.g., reduced number and distribution of sampling sites).

Table B-2. Population parameters for the Rio Grande Silvery Minnow, using October sampling-site data, for the Angostura Reach (1993–2021).

Year	N	*E(x)*	*E(x)* LCI	*E(x)* UCI	δ	μ
1993	5	10.44	0.85	128.05	1.00	0.76
1994	4	12.29	0.50	302.92	0.75	1.26
1995	5	1.28	0.55	3.02	1.00	-0.10
1996	6				0.17	-1.35
1997	5	7.62	3.75	15.48	1.00	1.77
1998	0					
1999	5				0.20	0.86
2000	5	0.00			0.00	
2001	5				0.20	-0.78
2002	5	0.26	0.07	0.93	0.60	-1.16
2003	5	0.00			0.00	
2004	5	2.17	0.44	10.80	0.80	0.28
2005	5	6.18	2.02	18.90	1.00	1.29
2006	5	0.73	0.51	1.06	1.00	-0.39
2007	5	22.89	9.64	54.36	1.00	2.77
2008	5	13.05	2.29	74.29	1.00	1.58
2009	5	15.34	3.75	62.71	0.80	2.37
2010	5	0.32	0.15	0.68	0.80	-1.08
2011	5	0.25	0.09	0.73	0.40	-0.47
2012	5	0.00			0.00	
2013	5	0.00			0.00	
2014	5	0.00			0.00	
2015	5	0.19	0.09	0.44	0.60	-1.18
2016	5	3.81	0.84	17.20	0.80	0.91
2017	10	13.85	7.24	26.50	0.90	2.40
2018	10	0.11	0.06	0.22	0.50	-1.51
2019	10	5.90	2.11	16.50	1.00	1.00
2020	10	0.07			0.10	-0.37
2021	10	0.37	0.12	1.15	0.50	-0.72

Sampling did not occur in 1998. $E(x)$ could not be estimated for 1996, 1999, 2001, or 2020; μ could not be estimated for 2000, 2003, 2012, 2013, or 2014 (i.e., all zero values).

Table B-3. Population parameters for the Rio Grande Silvery Minnow, using October sampling-site data, for the Isleta Reach (1993–2019).

Year	N	*E(x)*	*E(x)* LCI	*E(x)* UCI	δ	μ
1993	4	6.01	1.62	22.40	1.00	1.22
1994	4	4.94	1.17	20.88	0.75	1.42
1995	4	22.14	2.80	175.21	1.00	2.02
1996	4	0.67	0.23	1.89	0.75	-0.36
1997	2	6.07	0.52	70.44	1.00	0.96
1998	0					
1999	2				0.50	-0.05
2000	2	0.00			0.00	
2001	6	0.67	0.11	4.28	0.50	-0.36
2002	6				0.17	-1.99
2003	6				0.17	-1.36
2004	6	0.19	0.04	0.88	0.33	-0.80
2005	6	77.13	35.91	165.65	1.00	4.01
2006	6	0.65	0.27	1.56	0.83	-0.57
2007	6	19.05	3.16	114.85	1.00	1.78
2008	6	4.10	1.25	13.48	1.00	0.75
2009	6	6.62	3.73	11.73	1.00	1.68
2010	6	0.82	0.50	1.35	0.83	-0.09
2011	6	0.81	0.23	2.86	0.33	0.81
2012	6	0.00			0.00	
2013	6	0.00			0.00	
2014	6	0.00			0.00	
2015	6	0.26	0.12	0.56	0.67	-1.08
2016	6	8.79	0.88	88.21	1.00	0.58
2017	10	16.61	9.02	30.60	1.00	2.45
2018	10	0.06	0.01	0.23	0.20	-1.31
2019	10	3.35	1.65	6.82	0.90	0.93

Sampling did not occur in 1998. $E(x)$ could not be estimated for 1999, 2002, or 2003; μ could not be estimated for 2000, 2012, 2013, or 2014 (i.e., all zero values).

Table B-4. Population parameters for the Rio Grande Silvery Minnow, using October sampling-site data, for the San Acacia Reach (1993–2021).

Year	N	*E(x)*	*E(x)* LCI	*E(x)* UCI	δ	μ
1993	7	16.96	7.66	37.55	1.00	2.42
1994	5	24.39	7.06	84.23	0.60	3.40
1995	7	60.18	13.67	264.95	1.00	3.10
1996	5	3.84	1.67	8.85	1.00	1.01
1997	8	24.52	7.27	82.75	1.00	2.36
1998	0					
1999	8	12.71	4.70	34.34	1.00	1.91
2000	8	0.81	0.40	1.65	0.75	-0.14
2001	9	1.56	0.79	3.10	1.00	0.05
2002	9				0.11	-1.78
2003	9	0.00			0.00	
2004	9	0.71	0.17	2.92	0.33	0.42
2005	9	30.76	11.91	79.48	1.00	2.78
2006	9	1.42	0.48	4.17	0.89	-0.22
2007	9	1.71	0.51	5.73	0.56	0.64
2008	9	11.57	6.87	19.48	1.00	2.20
2009	9	19.49	9.81	38.71	1.00	2.57
2010	9	2.31	0.67	7.93	0.67	0.62
2011	9	2.71	0.31	23.54	0.44	0.76
2012	9	0.00			0.00	
2013	9	0.07	0.03	0.17	0.33	-1.62
2014	9	0.00			0.00	
2015	9				0.11	-0.47
2016	9	7.72	2.13	28.00	0.89	1.28
2017	10	27.04	10.66	68.64	1.00	2.62
2018	10	0.16	0.02	1.11	0.20	-0.66
2019	9	2.23	1.00	4.97	1.00	0.30
2020	9	0.07	0.02	0.26	0.22	-1.22
2021	10	0.52	0.25	1.12	0.70	-0.58

Sampling did not occur in 1998. $E(x)$ could not be estimated for 2002 or 2015; μ could not be estimated for 2003, 2012, or 2014 (i.e., all zero values).

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