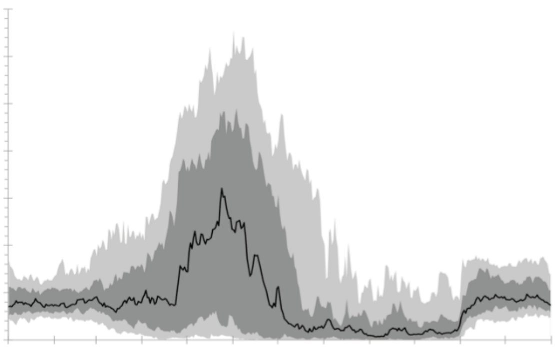
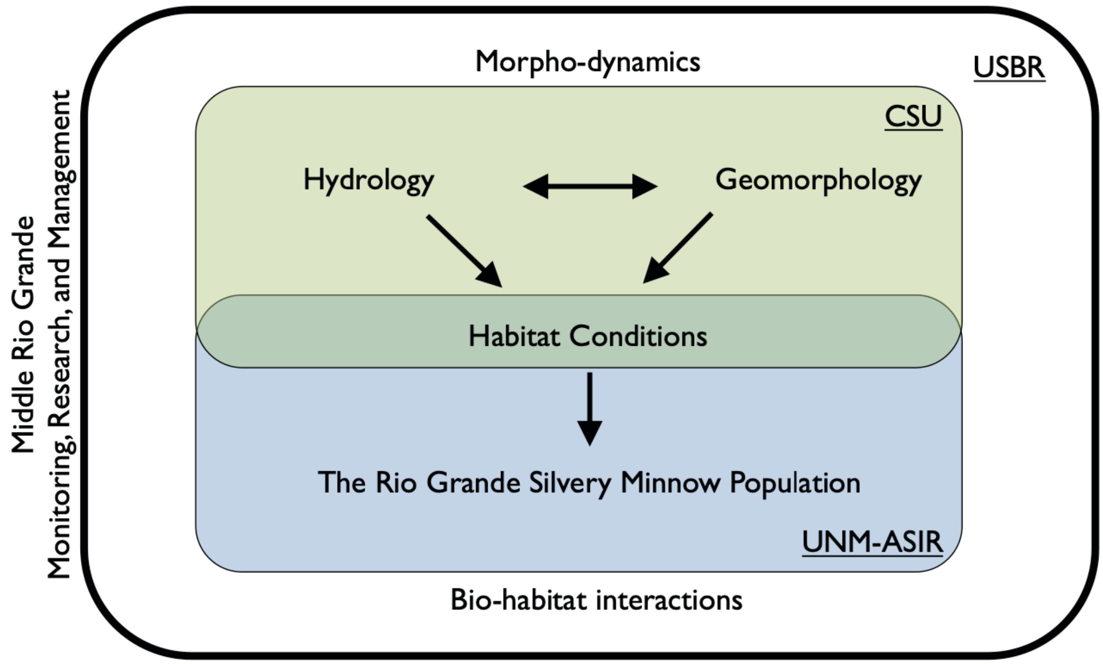
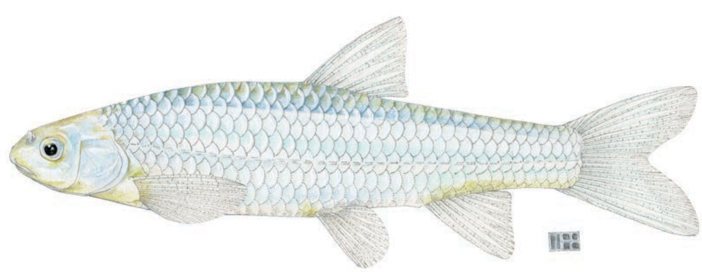
***Linking Morpho-Dynamics and Bio-Habitat Conditions on the Middle Rio Grande: Process-Linkage Report II – San Acacia Reach Analyses***

***Final Report***

***A U.S. Bureau of Reclamation Funded Research Project***





Final Report
23 March 2023

***LINKING MORPHO-DYNAMICS AND BIO-HABITAT CONDITIONS ON THE MIDDLE RIO GRANDE: PROCESS-LINKAGE REPORT II – SAN ACACIA REACH ANALYSES***
***FINAL REPORT***

***A U.S. BUREAU OF RECLAMATION FUNDED RESEARCH PROJECT***

**U.S. Bureau of Reclamation**
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Submitted to:

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## EXECUTIVE SUMMARY

This report (the 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, L.L.C.

The principal objectives of this study were to:

- Identify and assess 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 suspected 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 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 San Acacia 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.

The main findings of this study included:

- Key process-linkages identified for the San Acacia Reach 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 San Acacia Reach showed distinct, spatially variable trends over time (1962–2012) that differed considerably from the Isleta Reach (Linkage Report I).
- Discharge was consistently lower in the San Acacia Reach compared to the Isleta Reach (including increased frequency of intermittency), however, habitat metrics were consistently greater during the study period (1993–2021).
- Channel aggradation was prevalent downstream of Escondida, which corresponded to floodplain connectivity and greater larval habitat availability – water surface elevation at Elephant Butte Reservoir was shown to control morpho-dynamics in the downstream-most subreaches over time.
- Densities of the Rio Grande Silvery Minnow were generally higher in the San Acacia Reach relative to the Isleta Reach. Higher densities were attributed to greater larval habitat availability and pertinent ecological processes (i.e., downstream drift/dispersal).
- Flow and habitat metrics corresponding to the larval life-stage of Rio Grande Silvery Minnow were the most reliable long-term predictors of the species' density and occurrence at the reach-scale, however, flow metrics explained more variation in population parameters across years.
- Habitat metrics showed that dramatic increases in larval habitat availability (i.e., several orders of magnitude) were linked to prolonged overbanking flows.
- 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. Current limitations to hydraulic modeling and habitat analyses include the estimation of overbanking discharges for perched/semi-perched channels and limited accuracy of modeling low flows.
- Flow management in the San Acacia Reach will be important to the recovery of the Rio Grande Silvery Minnow. Relatively low overbanking discharges and floodplain connectivity downstream of Escondida suggest abundant larval habitats can be created given sufficient spring runoff, however, high frequency of intermittency during the summer is detrimental to survival. Restoration of larval habitats is expected to be most effective between San Acacia Diversion Dam and Escondida due to high channel incision, perennial flows, and upstream location.

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# INTRODUCTION

*Purpose and Objectives*

This report (the 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 Linkage Report are to:

- Identify and assess 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 suspected 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 second Linkage Report produced for this project, which includes an assessment of process-linkages for the San Acacia Reach of the Middle Rio Grande (San Acacia Diversion Dam to San Marcial, NM) – the first Linkage Report targeted the Isleta Reach (Isleta Diversion Dam to San Acacia Diversion Dam). Insights gained from the Isleta Reach included:

- Key process-linkages identified were (1) floodplain connectivity and inundation, and (2) main channel habitat complexity and availability
- Reach geomorphology was generally characterized by channel narrowing and incision over time caused by reduced sediment supply, channelization, and riparian vegetation
- Geomorphic trends 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 (1993–2019)

Future Linkage Reports will incorporate additional reach analyses and monitoring data as they become available (i.e., Angostura/Albuquerque Reach). This report reflects the input of numerous collaborators over several years to refine the methods, analyses, and results. As this project continues to evolve, it is possible that sections of this report may be modified for future reports based on input from collaborators, forthcoming analyses, and project development.

*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 (USOFR, 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 15 years, 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 Ichythyological 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., 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 river 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 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.

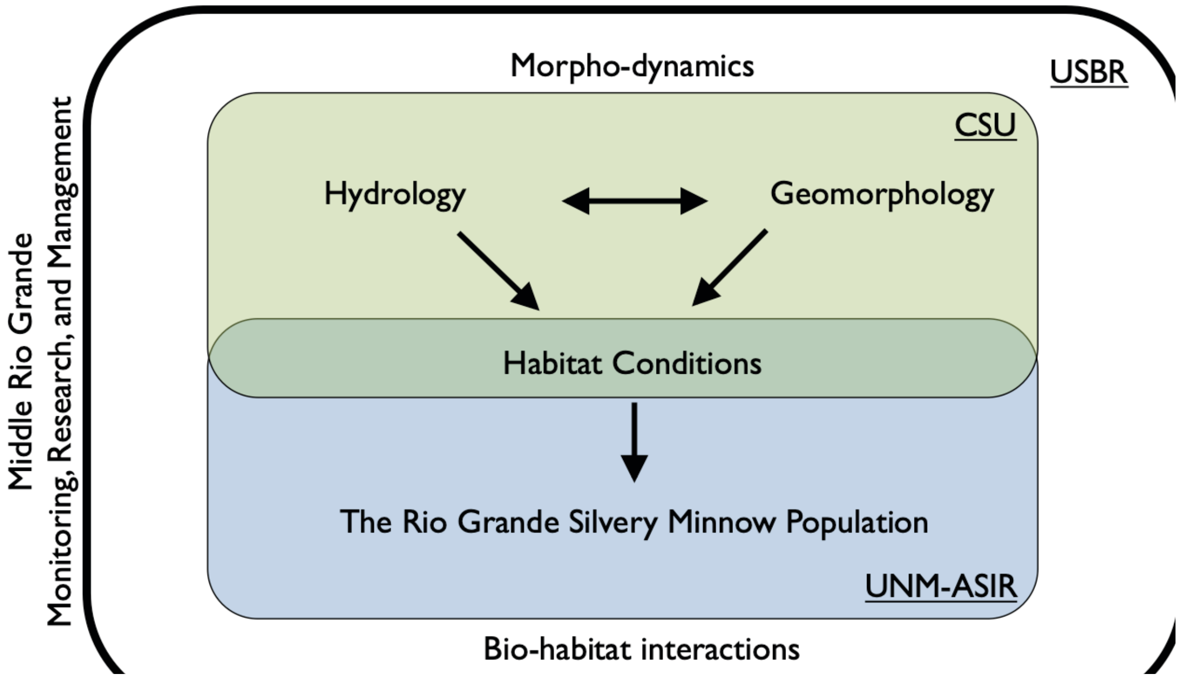



Figure 1. Simplified conceptual diagram of linkages among morpho-dynamic processes and bio-habitat interactions in the Middle Rio Grande. Boxes represent the knowledge and expertise of scientific disciplines and project collaborators.

*Linking Morpho-dynamics and Bio-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, <u>RE</u>storing rivers <u>FOR</u> effective catchment <u>M</u>anagement (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.

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 characterized and assessed 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 (USOFR, 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 (USOFR, 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 San Marcial, NM (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 II – San Acacia Reach*

This Linkage Report focuses on the San Acacia Reach of the Middle Rio Grande, which is defined herein as the length of river between San Acacia Diversion Dam and San Marcial, NM (57.1 mi [91.9 km]; Figure 3). The Reach Reports analyzed this reach in four segments based on previous USBR designations: (1) the “San Acacia (SA) Reach” between San Acacia Diversion Dam and the Escondida Bridge (Doidge et al., 2020), (2) the “Escondida (E) Reach” between the Escondida Bridge and the US380 Bridge (Beckwith and Julien, 2020), (3) the “Bosque del Apache (BDA) Reach” between the US380 Bridge and the southern boundary of Bosque del Apache National Wildlife Refuge (BDA-NWR; Schied et al., 2022), and (4) the “Elephant Butte (EB) Reach” between the southern boundary of BDA-NWR and the inflow to Elephant Butte reservoir (Sperry et al., 2022). Due to variation in pool elevation of Elephant Butte reservoir during the study period, the downstream boundary of the study area was designated as the terminus of the low flow conveyance channel (agg/deg line 1794; near San Marcial, NM) for purposes herein. Subreaches were delineated in the Reach Reports based on geomorphic characteristics (e.g., SA1–SA4, E1–E5) – these designations were kept for this Linkage Report (Figure 3, Table 1). Subreach delineation methods are described in Data and Methods. For clarity, this Linkage Report defines the upstream and downstream boundaries of the San Acacia Reach as San Acacia Diversion Dam and the terminus of the low flow conveyance channel (agg/deg line 1794) near San Marcial, NM.

Linkage Report I targeted the Isleta Reach (Isleta Diversion Dam to San Acacia Diversion Dam) and Linkage Report III will target the Angostura Reach (Angostura Diversion Dam to Isleta Diversion Dam). Insights and knowledge gained from Linkage Report I informed the preparation of Linkage Report II — future results and analyses will be incorporated into Linkage Report III following completion of Reach Reports between Angostura Diversion Dam and Isleta Diversion Dam.

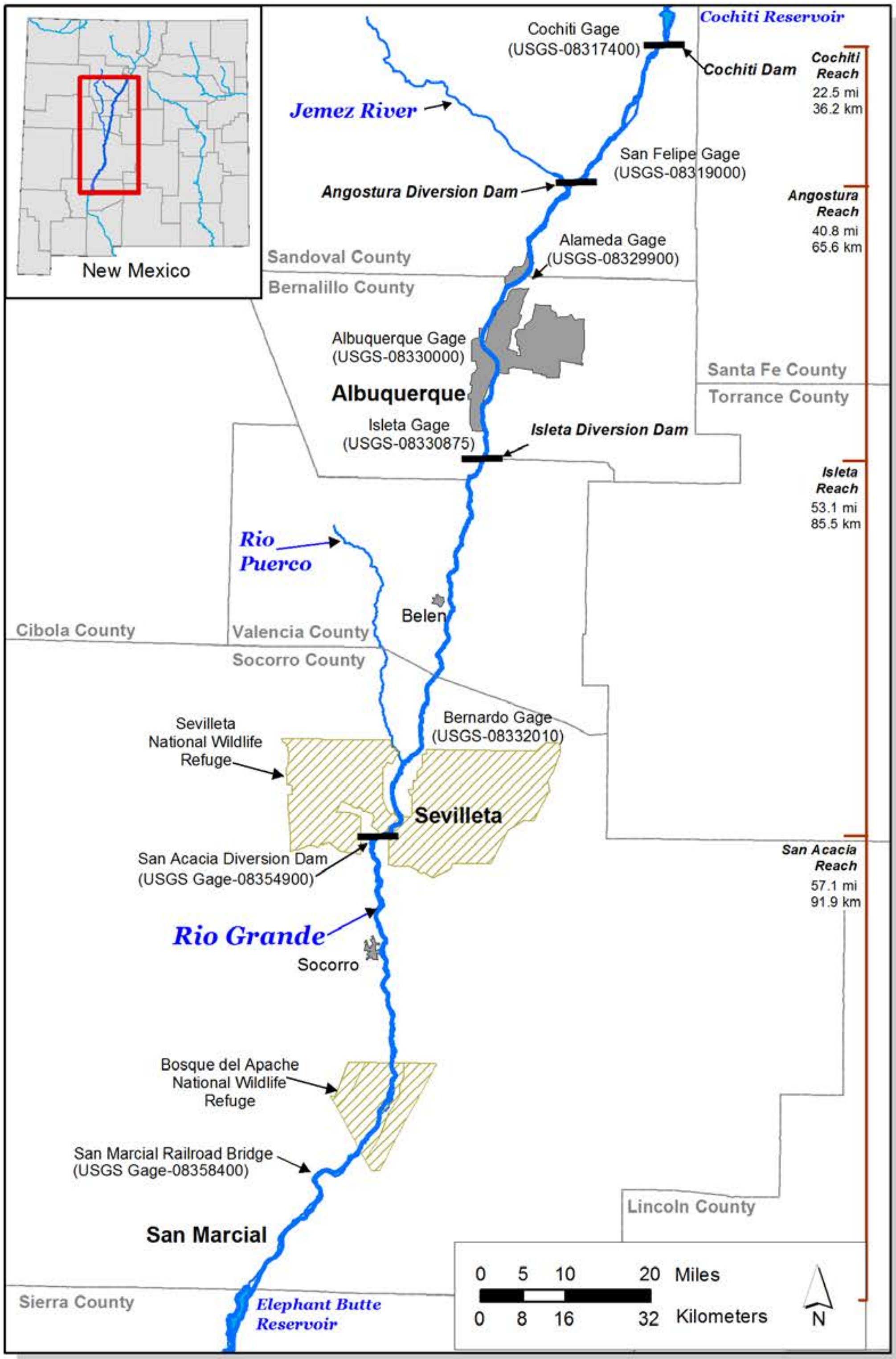



Figure 2. Map of the Middle Rio Grande, NM with selected features. Reaches between diversion dams (e.g., San Acacia Reach) are shown at right.

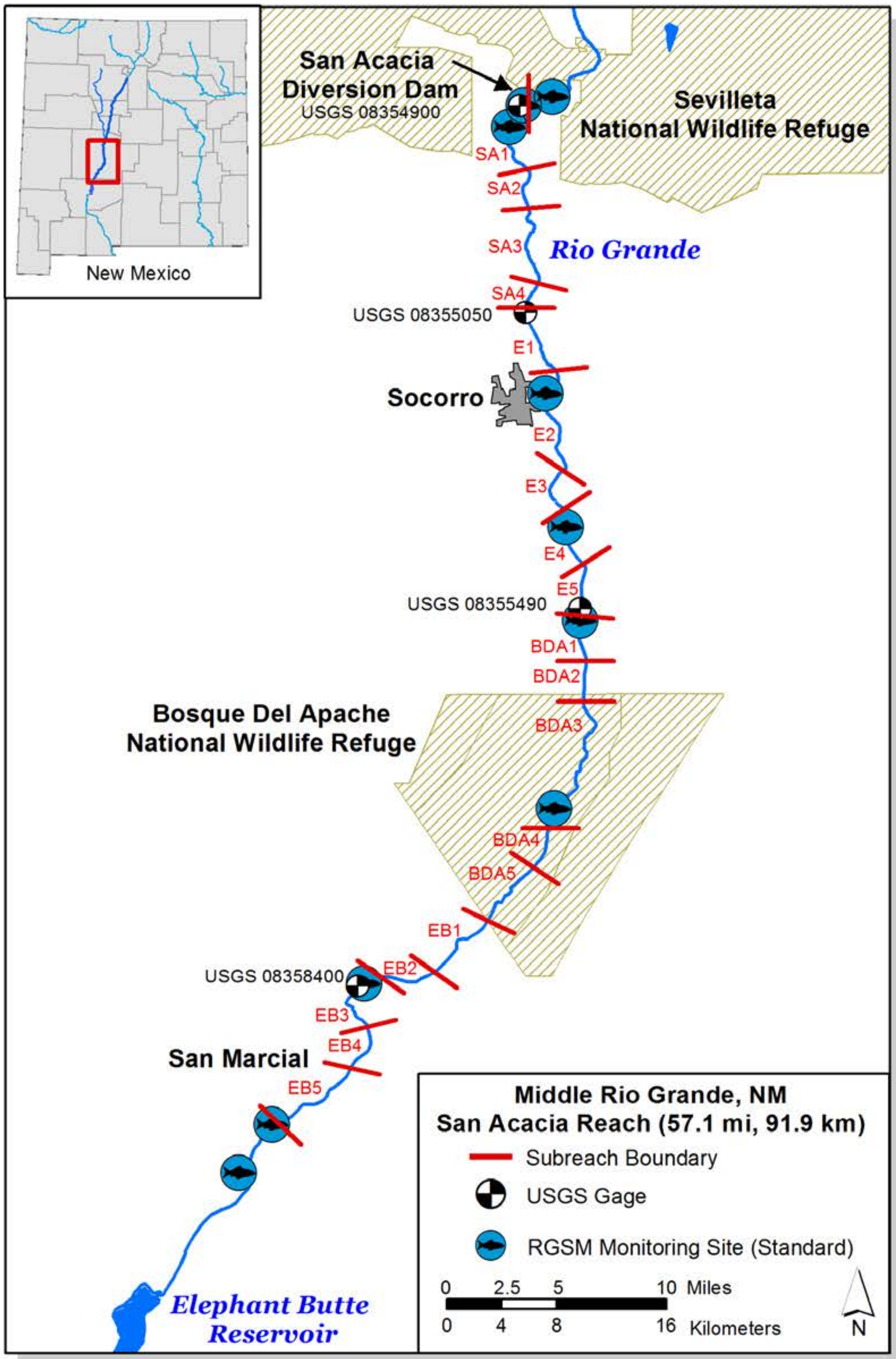



Figure 3. Map of the San Acacia Reach (San Acacia Diversion Dam to San Marcial, NM) of the Middle Rio Grande with selected features.

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

| Subreach | Agg/deg Line | (Rangeline) | Length, mi | (km) | Geomorphic Features |
|---|---|---|---|---|---|
| SA1 | 1207–1245 | (741–704) | 3.8 | (6.1) | San Acacia Diversion Dam |
| SA2 | 1245–1264 | (704–685) | 2.0 | (3.3) | Confluence with Alamillo Arroyo |
| SA3 | 1264–1300 | (685–649) | 3.7 | (6.0) | Change in cumulative depth (shallower) |
| SA4 | 1300–1313 | (649–636) | 1.3 | (2.1) | Change in cumulative width (narrower) |
| E1 | 1313–1345 | (636–604) | 3.2 | (5.1) | Escondida Bridge |
| E2 | 1345–1397 | (604–552) | 5.0 | (8.0) | Low radius bend |
| E3 | 1397–1420 | (552–530) | 2.1 | (3.4) | Confluence with Arroyo De Las Cañas |
| E4 | 1420–1448 | (530–502) | 2.7 | (4.3) | Change in cumulative width (wider) |
| E5 | 1448–1475 | (502–475) | 2.5 | (4.0) | Change in cumulative width (narrower) |
| BDA1 | 1475–1496 | (475–454) | 2.1 | (3.3) | US380 Bridge near San Antonio, NM |
| BDA2 | 1496–1516 | (454–434) | 1.9 | (3.1) | Change in cumulative depth (shallower) |
| BDA3 | 1516–1582 | (434–369) | 6.6 | (10.7) | Change in cumulative width (wider) |
| BDA4 | 1582–1603 | (369–350) | 2.2 | (3.5) | Change in cumulative width (narrower) |
| BDA5 | 1603–1637 | (350–319) | 3.0 | (4.9) | Change in cumulative width (narrower) |
| EB1 | 1637–1672 | (319–285) | 3.2 | (5.1) | Southern boundary of BDA NWR |
| EB2 | 1672–1696 | (285–262) | 2.3 | (3.7) | Change in cumulative width (narrower) |
| EB3 | 1696–1728 | (262–232) | 3.1 | (5.0) | Change in cumulative width (wider) |
| EB4 | 1728–1751 | (232–210) | 2.1 | (3.4) | Change in cumulative width (narrower) |
| EB5 | 1751–1794 | (210–169) | 4.3 | (6.8) | Change in cumulative width (wider) |

### *Geomorphic and Hydrologic Characteristics – San Acacia Reach Summary*

Previous reach reports described the channel profiles for the San Acacia (SA), Escondida (E), Bosque del Apache (BDA), and Elephant Butte (EB) subreaches of the San Acacia Reach of the Middle Rio Grande (Doidge et al., 2020, Beckwith et al., Schied et al., 2022, Sperry et al., 2022). A longitudinal profile between the San Acacia Diversion Dam and Elephant Butte Reservoir was developed to illustrate reach-scale trends of degradation and aggradation over time (Figure 4). Overall, the channel slope of the reach between San Acacia and Elephant Butte has fluctuated between 0.06% and 0.09% between 1962 and 2012.

The San Acacia subreach at the upstream end of the reach was steepest in 1962, with an average bed slope of 0.09%. By 2012, this subreach incised and flattened to an average slope of 0.08%. The subreach experienced between 3 feet and 10.5 feet of degradation between 1962 and 2012, with the most degradation occurring at the San Acacia Diversion Dam. This long-term trend of channel incision is likely the result of sediment containment behind the San Acacia Diversion Dam. (Doidge et al., 2020). This subreach is also influenced by uplift from the Socorro Magma Body (e.g., Figure 14).

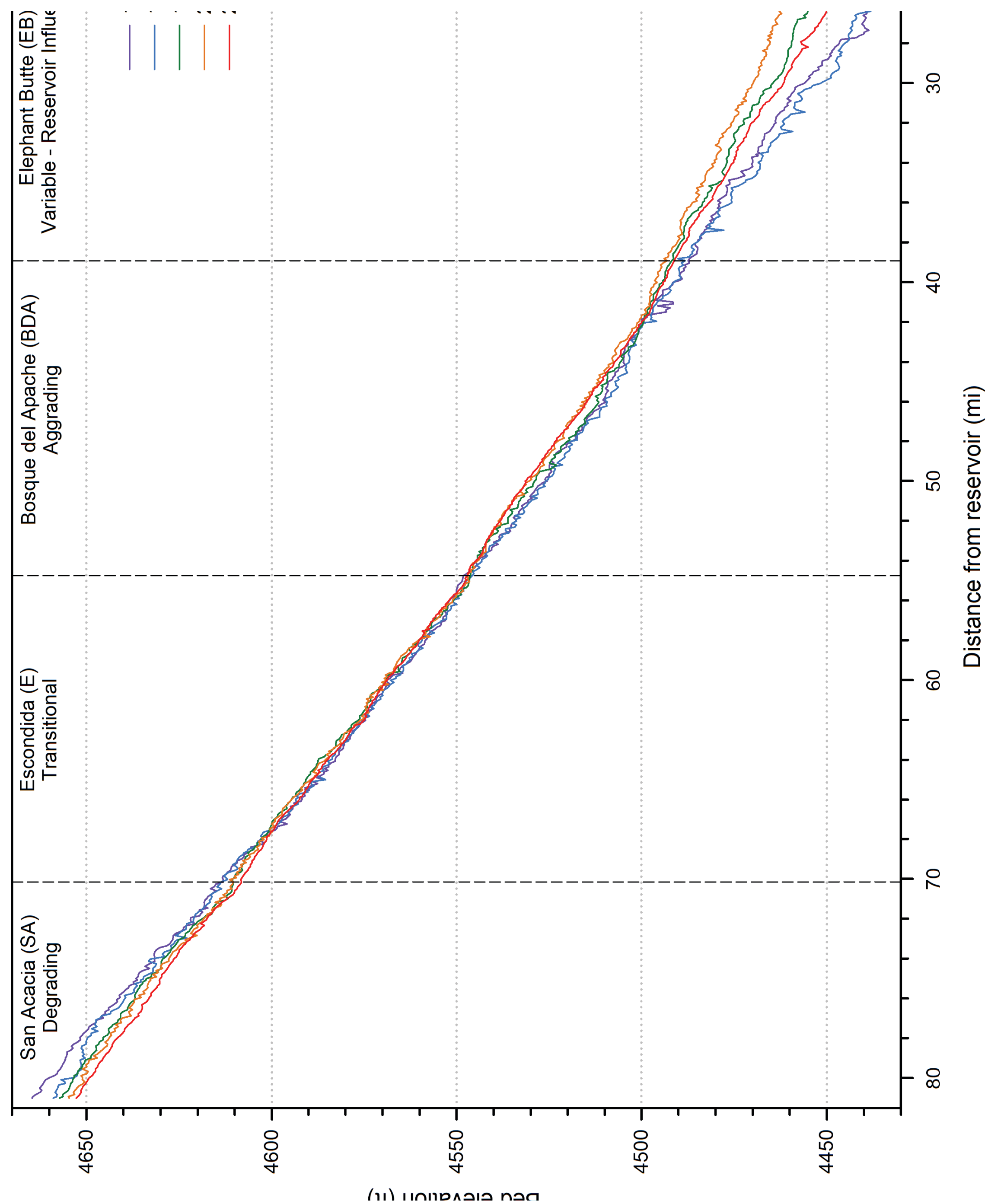
Channel incision terminates approximately 13 miles downstream of the San Acacia Diversion Dam in the Escondida subreach, where the river appears to stabilize at a pivot point located approximately 68 miles upstream of Elephant Butte Reservoir (Beckwith et al., 2020). Downstream of this point, the channel bed in the Escondida subreach experienced fluctuations during the 50 years of record, but the trends of aggradation and degradation are not as pronounced as those observed in other subreaches.

The Bosque del Apache subreach experienced up to 5 feet of aggradation between 1972 and 1992 (Schied et al., 2022). It appears that there is a second pivot point within the Bosque del Apache subreach at a location 41.7 miles upstream of Elephant Butte Reservoir. At this point, the channel has not experienced large magnitude change to bed elevations during the 50 years of record, and the 2012 bed profile shows a hinge point where the overall channel slope shifts slightly from 0.08% to 0.06%.

Within the Elephant Butte subreach, considerable aggradation occurred due to elevated reservoir levels at Elephant Butte Reservoir between 1972 and 2002. At the downstream end of the subreach, the channel aggraded up to 22 feet. This is followed by a period of considerable degradation between 2002 and 2012. At the downstream end of the subreach, the channel degraded up to 13 feet. The aggradation and subsequent degradation of the channel correspond with large shifts in water levels at the reservoir. Between 1972 and 2002, the reservoir water level increased by around 120 feet, and after 2002, the water level decreased by about 90 feet (Sperry et al., 2022). The influence of reservoir level on channel bed elevation in this subreach has caused the evolutionary trajectory of this segment of river to diverge from those upstream and the magnitude of change in bed elevation has been greater as well.

Distinct spatial hydrologic trends are also evident within the San Acacia Reach. While annual and seasonal discharge trends are similar throughout this reach and the Middle Rio Grande, flows are generally lowest in the San Acacia Reach and tend to decrease in magnitude downstream of San Acacia Diversion Dam. Much of the San Acacia Reach is characterized as a losing reach. While the San Acacia subreach can be largely characterized as perennial, the three downstream subreaches often experience flow intermittency during the irrigation season (March–October). Additionally, the Low Flow Conveyance Channel (LFCC) was operational ca. 1955–1985 and was used to increase water conveyance to Elephant Butte Reservoir by bypassing the river channel at low flows. The LFCC is no longer used for its intended purpose but still functions as an agricultural drain. Perched and semi-perched channel conditions downstream of Escondida, NM heighten water losses from the river channel to the LFCC. Water management practices in this reach have included pumping water from the LFCC back to the river channel to attempt to maintain at least some flow in the river channel for fish habitat. The effects of low flows and flow intermittency on habitat availability and fish survival are discussed in further detail in this report.

Detailed subreach analyses, including flow and sediment trends, are presented in the respective reach reports, which were prepared as part of this study (Doidge et al., 2020, Beckwith et al., Schied et al., 2022, Sperry et al., 2022); a subset of these results are provided in Appendix C.


San Acacia (SA)
Degrading
Escondida (E)
Transitional
Bosque del Apache (BDA)
Aggrading
Elephant Butte (EB)
Distance from reservoir (mi)
30
40
50
60
70
80
4650
4600
4550
4500
4450

## *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 bank 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 [1959]). 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 MRGP 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.

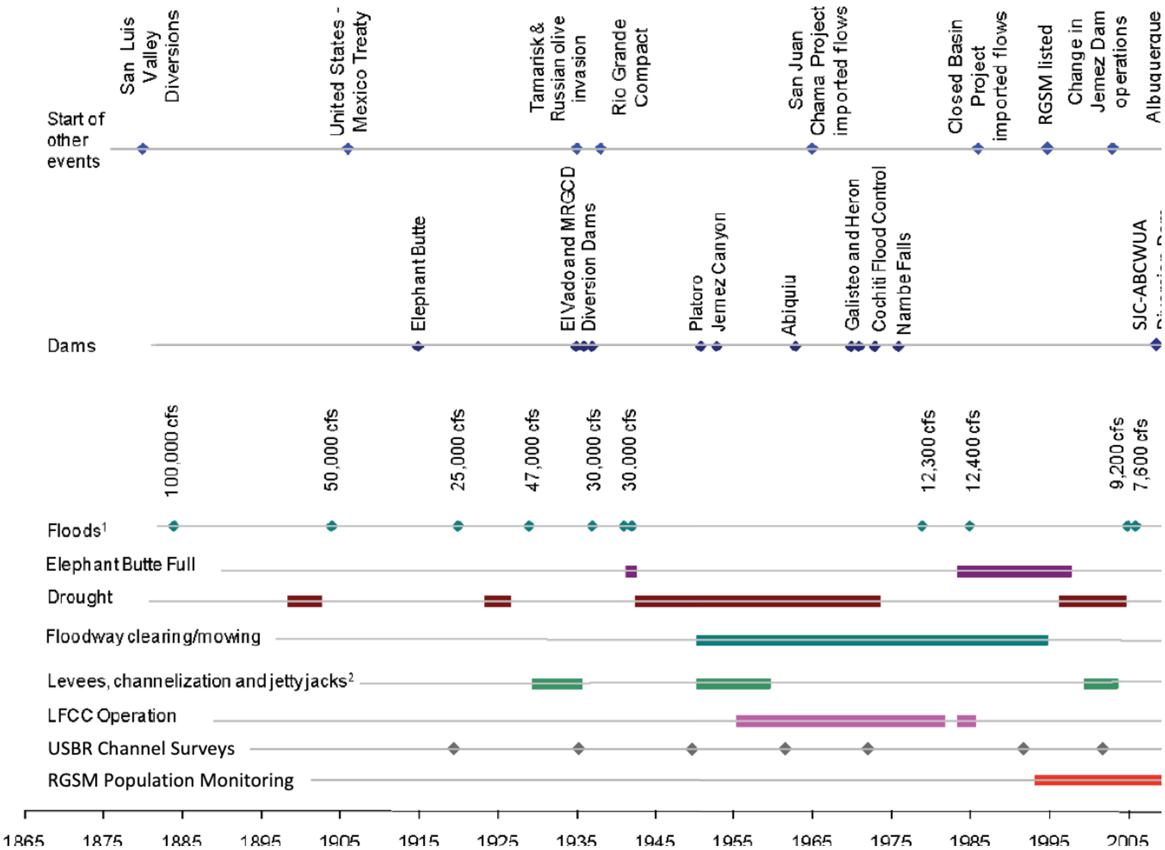



Figure 5. Timeline of substantial hydrologic and geomorphic impacts to the Middle Rio Grande, 1865–2015. Figure modified from Makar and Aubuchon (2012).

### *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.

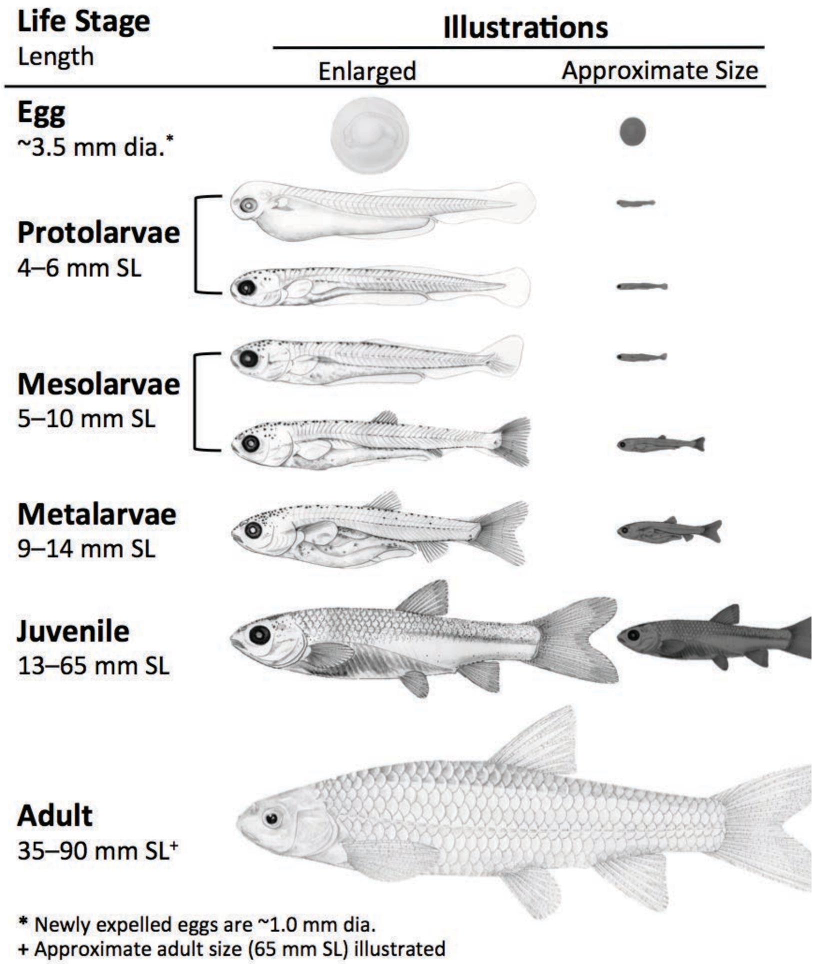
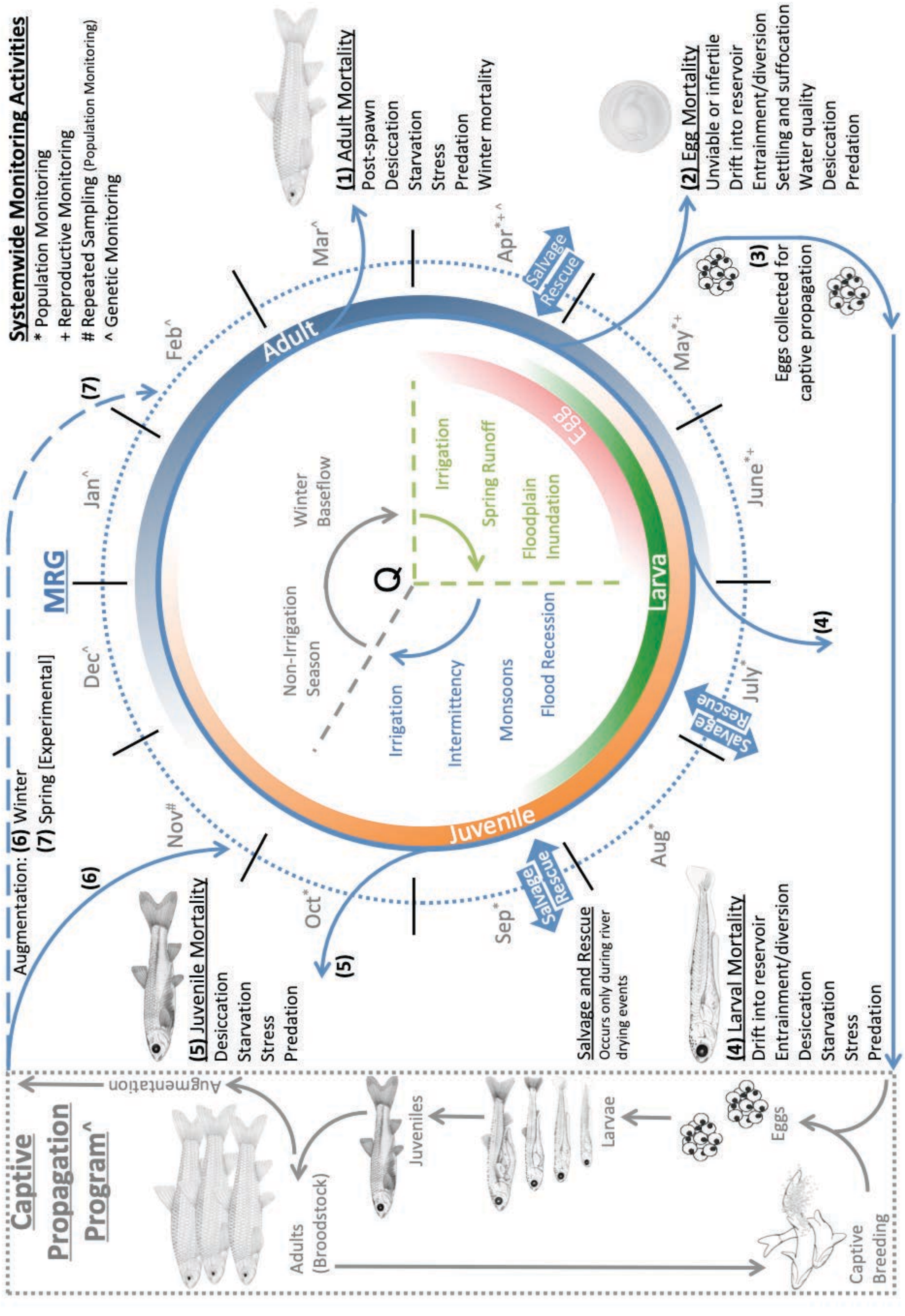
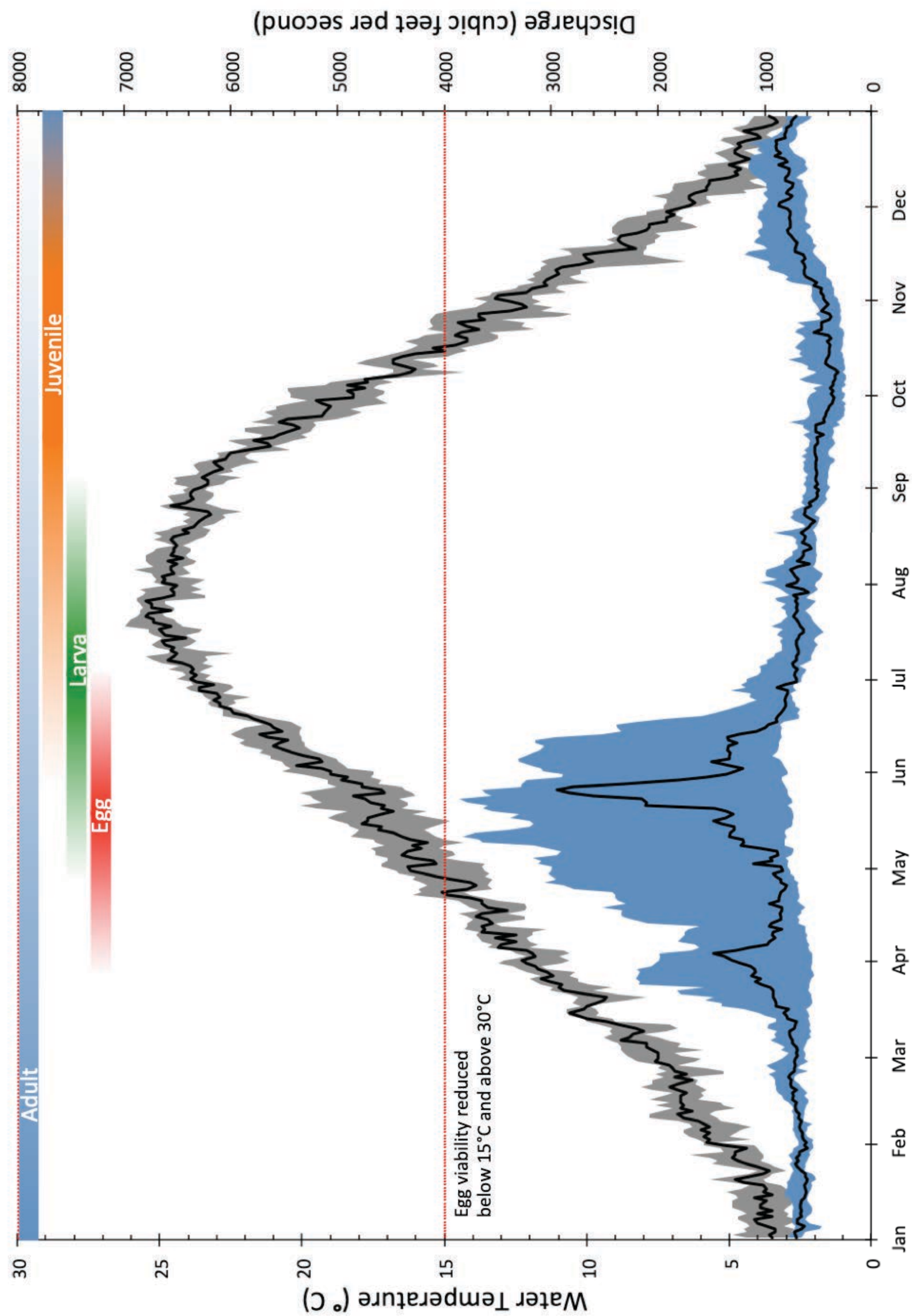
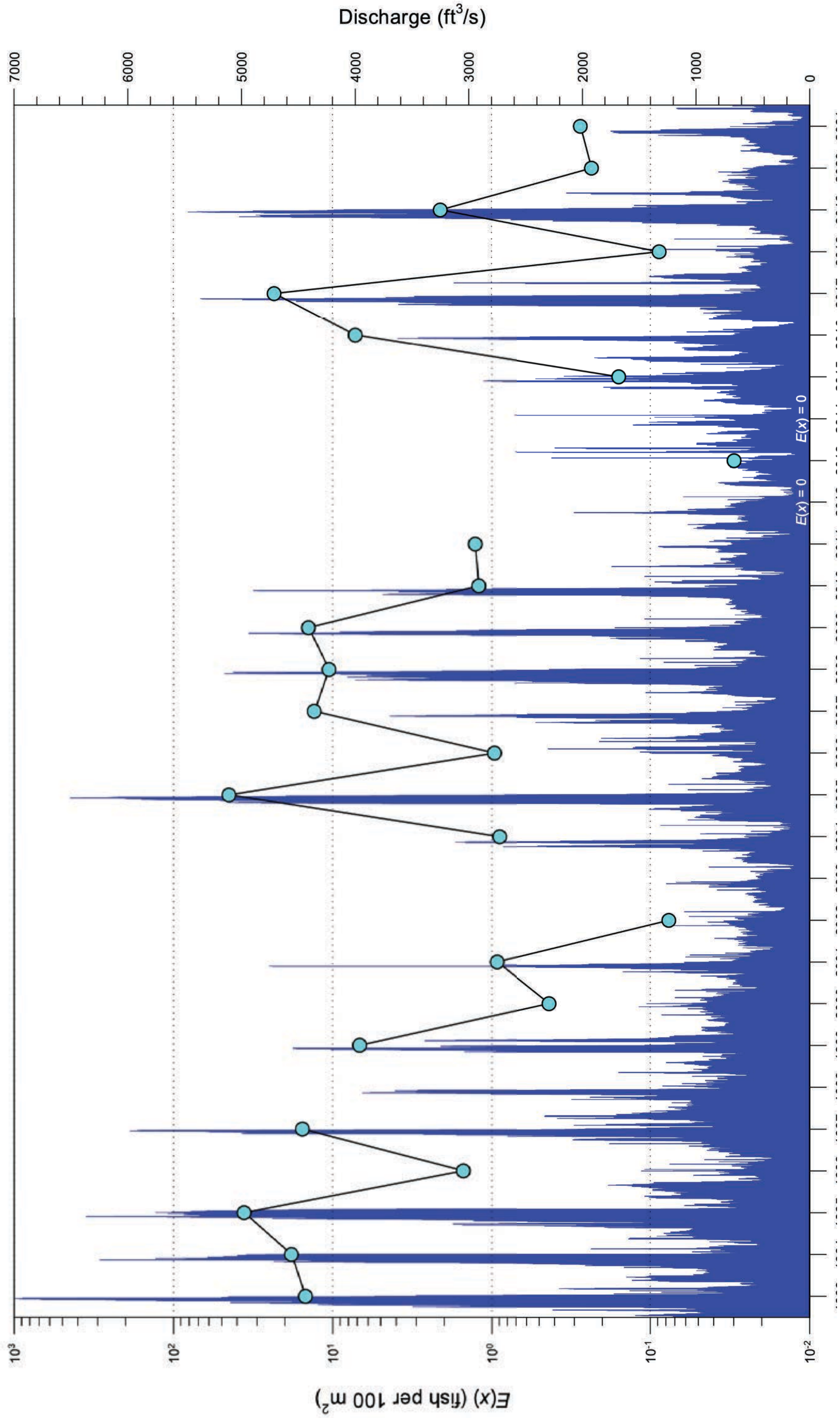



Figure 6. Life-stages of the Rio Grande Silvery Minnow and their approximate actual sizes. Lengths were obtained from Brandenburg et al. 2018 (egg, larvae, juvenile); USFWS 2010 and Horwitz et al. 2018 (juvenile, adult). Note considerable overlap of lengths especially between juvenile and adult. Illustrations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


Systemwide Monitoring Activities
* Population Monitoring
+ Reproductive Monitoring
# Repeated Sampling (Population Monitoring)
^ Genetic Monitoring
(1) Adult Mortality
Post-spawn
Desiccation
Starvation
Stress
Predation
Winter mortality
(2) Egg Mortality
Unviable or infertile
Drift into reservoir
Entrainment/diversion
Settling and suffocation
Water quality
Desiccation
Predation
(3)
Eggs collected for captive propagation
(4)
(5)
(6)
(7)
Augmentation: (6) Winter
(7) Spring [Experimental]
MRG
Jan^
Feb^
Mar^
Apr*+^
May*+
June*+
July*
Aug*
Sep*
Oct*
Nov#
Dec^
Adult
Egg
Larva
Juvenile
Salvage Rescue
Q
Winter Baseflow
Irrigation
Spring Runoff
Floodplain Inundation
Non-Irrigation Season
Irrigation
Intermittency
Monsoons
Flood Recession
(5) Juvenile Mortality
Desiccation
Starvation
Stress
Predation
Salvage and Rescue
Occurs only during river drying events
(4) Larval Mortality
Drift into reservoir
Entrainment/diversion
Desiccation
Starvation
Stress
Predation
Captive Propagation Program^
Augmentation
Adults (Broodstock)
Juveniles
Larvae
Eggs
Captive Breeding


Discharge (cubic feet per second)
8000
7000
6000
5000
4000
3000
2000
1000
0
Water Temperature (° C)
30
25
20
15
10
5
0
Adult
Juvenile
Larva
Egg
Egg viability reduced
below 15°C and above 30°C
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec


Discharge (ft$^3$/s)
7000
6000
5000
4000
3000
2000
1000
0
$E(x)$ = 0
$E(x)$ = 0
$10^3$
$10^2$
$10^1$
$10^0$
$10^{-1}$
$10^{-2}$
$E(x)$ (fish per 100 m$^2$)

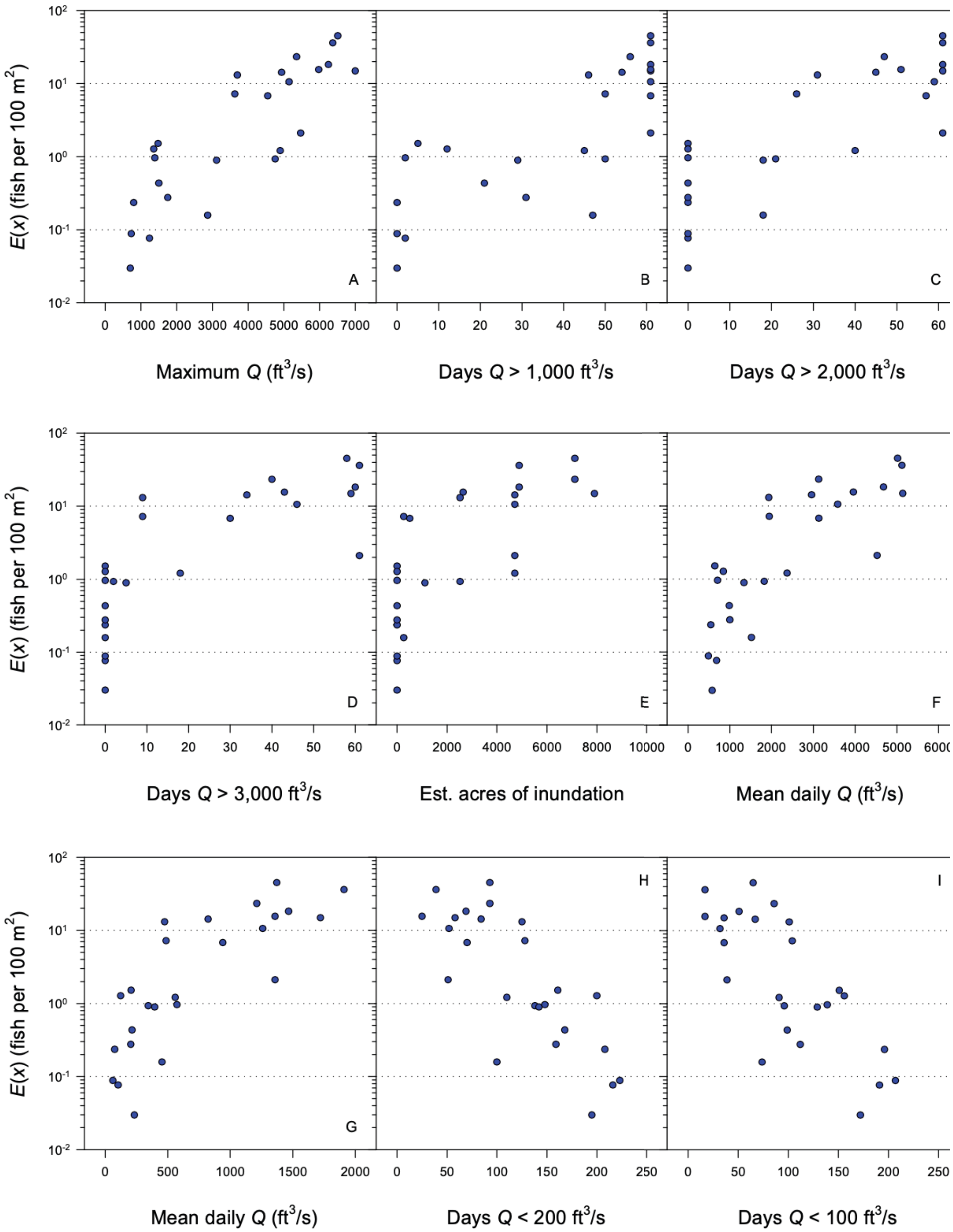



Figure 10. Bivariate plots of densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data) and seasonal hydrologic metrics 1993–2021 (Albuquerque Gage USGS 08330000 data [A–F], and San Marcial Gage USGS 08354900 data [G–I]; from Dudley et al. 2022).

*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; Klein et al., 2018). 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; Klein et al., 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 has 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; Klein et al., 2018). Large-scale channelization in the 1950s involved the installation of jetty jacks to stabilize banks and protect levees (Figure 13; 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).

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

[1] Channel degradation occurs in 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 drought periods, however, increased roughness on vegetated surfaces might result in marginal increases in the availability of slackwater habitats.
[4] Implications for population dynamics of the Rio Grande Silvery Minnow cumulatively contribute to reduced occurrence, abundance, and persistence in the Middle Rio Grande.

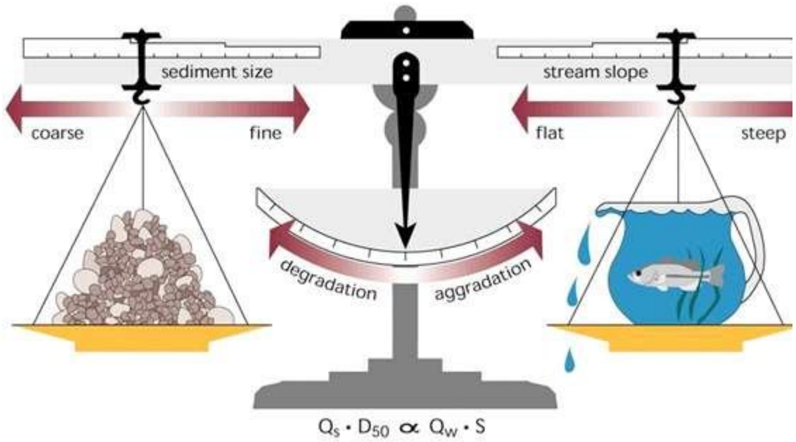



Figure 11. Illustration of Lane's Balance showing the relationship between sediment discharge ($Q_s$), median sediment size ($D_{50}$), water discharge ($Q_w$), channel slope (S), and aggradational or degradational responses to disturbance. Relationship developed by Lane (1955) and enhanced graphics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

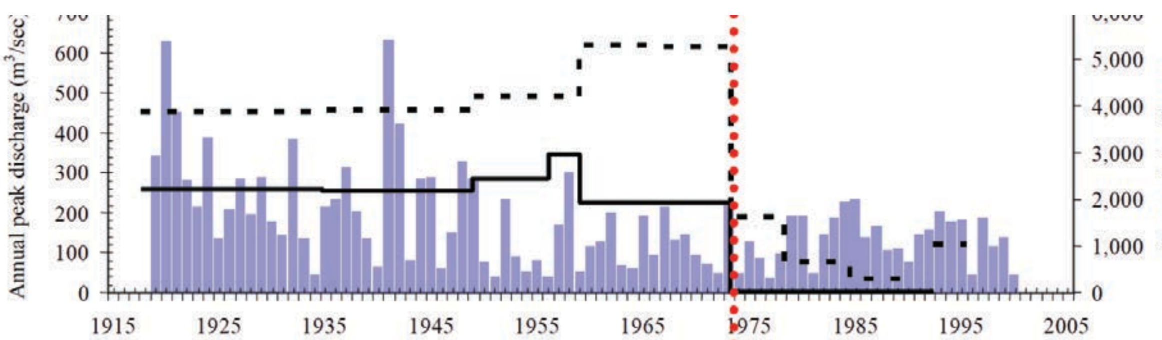



Figure 12. Summary of changes to flow and sediment regimes in the Middle Rio Grande 1918–2000 (from Richard and Julien, 2003). Construction of Cochiti Dam is indicated by the red dotted line (1973). Note higher peak flows and suspended sediment concentrations prior to closure of Cochiti Dam.


Figure 13. Example of channel modifications in the Isleta Reach downstream of Los Lunas, NM. Orange lines show jetty jack lines (1962) and the red line shows subreach boundaries. Locations of levees and riverside drains are visible. Left panel (I2) is just upstream of the right panel (I2–I3); each panel shows about 5 river miles (8 rkm). Example jetty jack (left) and line placement (right) are illustrated at bottom (Grassel, 2002). Aerial imagery obtained from GoogleEarth.

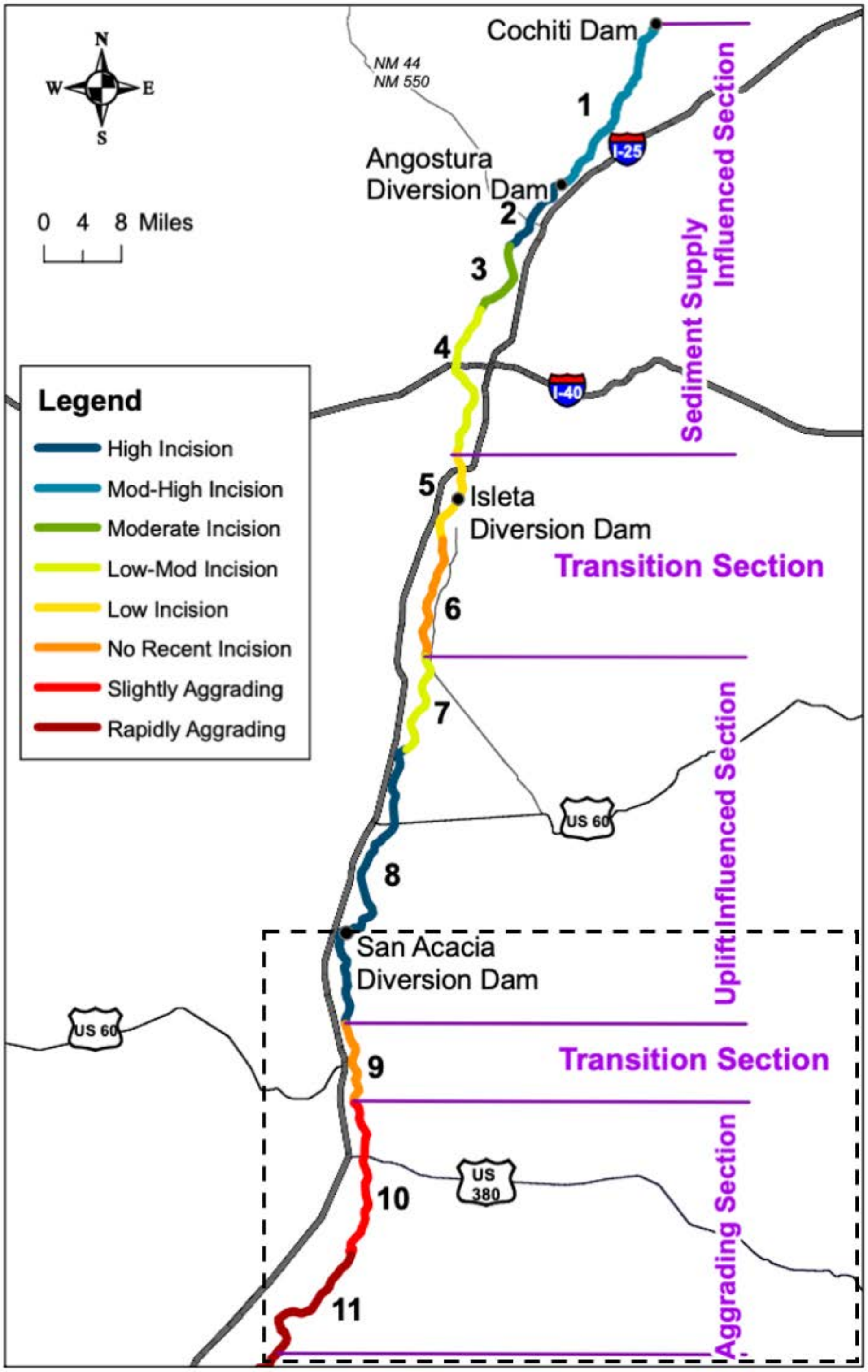



Figure 14. Middle Rio Grande aggradation-degradation trends 1936–2002 (from Massong et al. 2006). Trends are described spatially as: high incision (>6 ft), moderate incision (3–5 ft), low incision (<3 ft), slightly aggrading (<10 ft), and rapidly aggrading (>10 ft). Dashed black line shows the San Acacia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

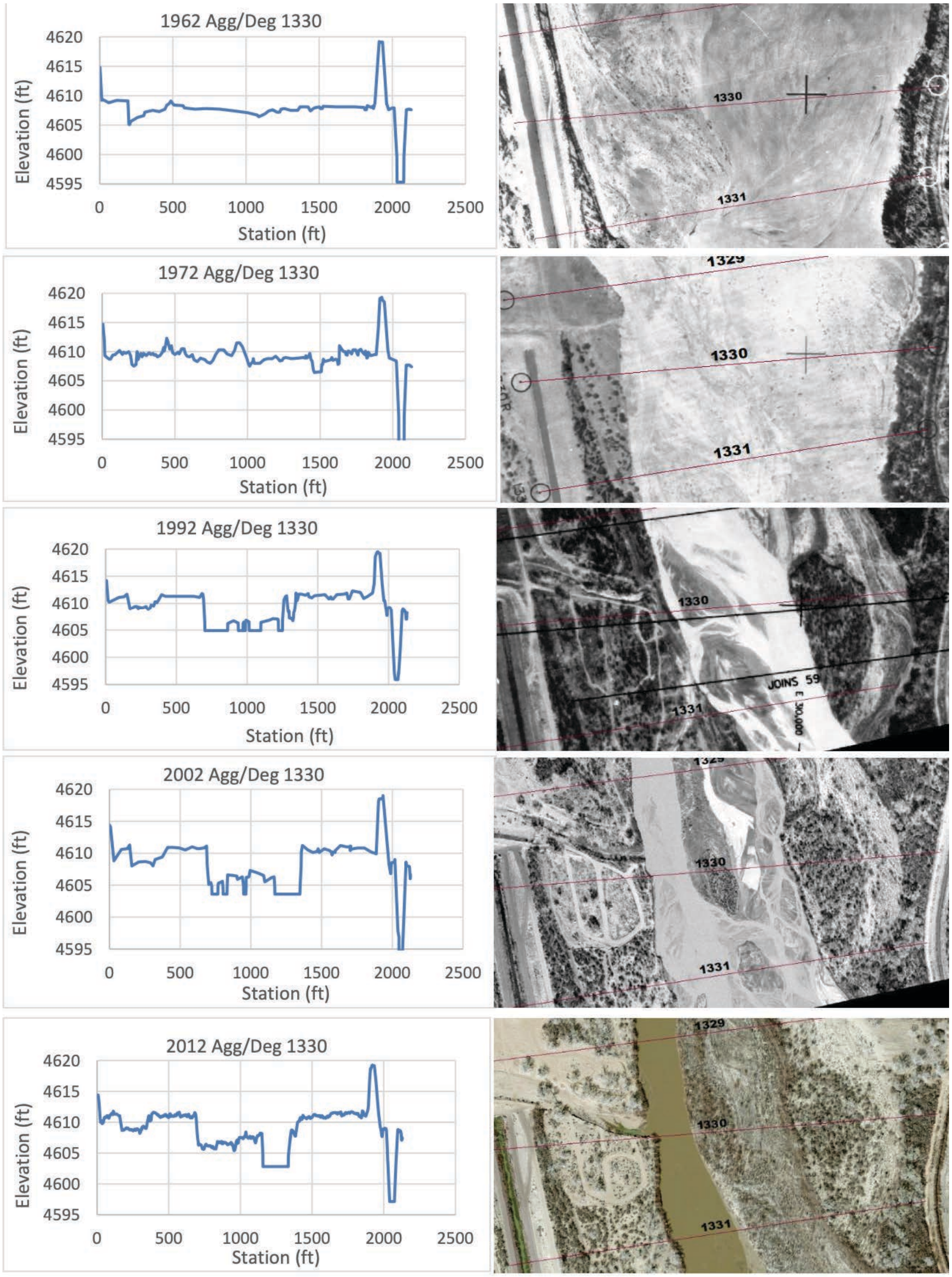



Figure 15. Example of channel narrowing and incision in the San Acacia Reach (from Beckwith and Julien, 2020). Panels (top to bottom) illustrate changes to the cross-section (left) and planform (right) at agg/deg line 1330 (subreach E1; 1.6 mi downstream of Escondida bridge) between 1962 and 2012.

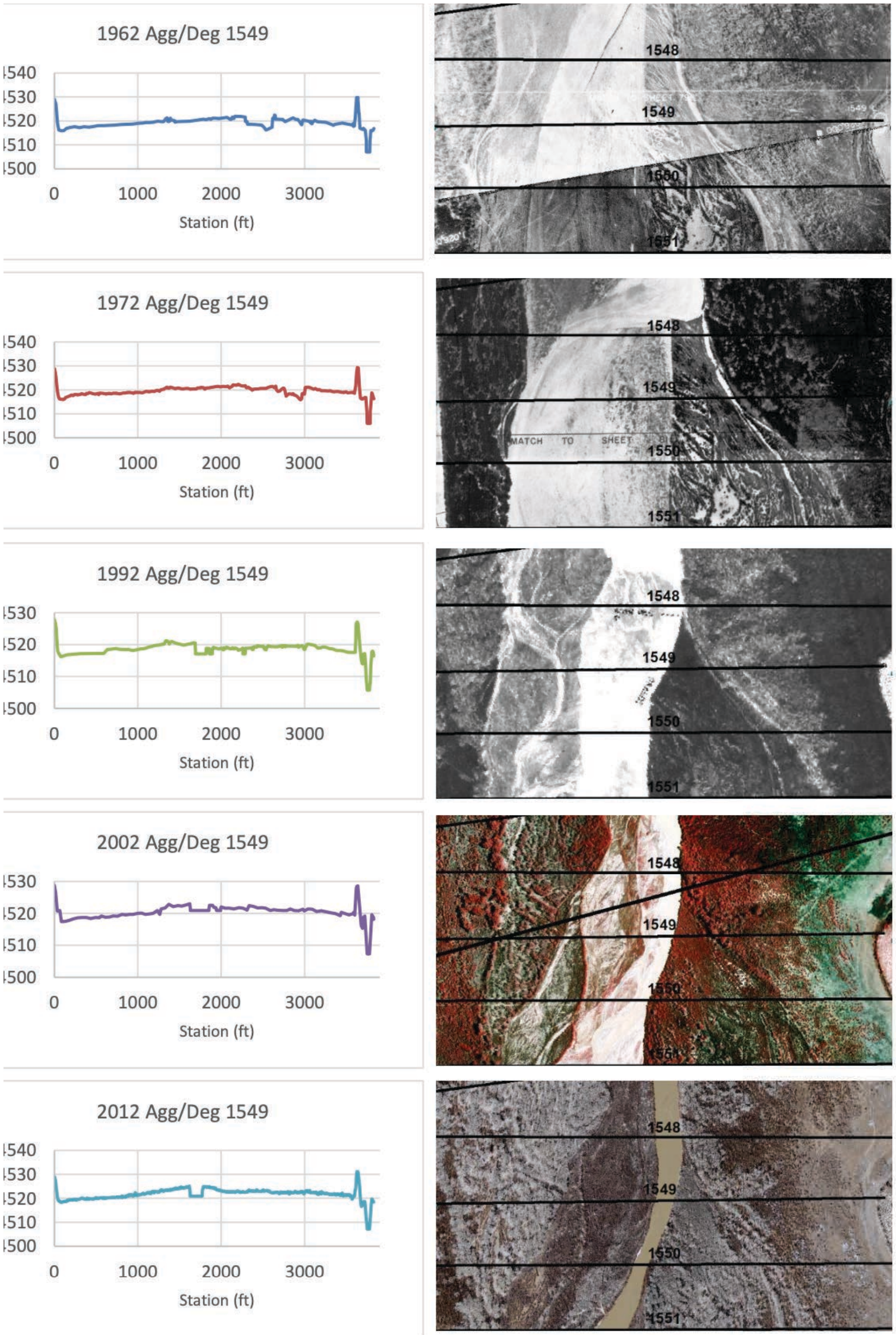



Figure 16. Example of channel narrowing and aggradation in the San Acacia Reach (from Schied et al., 2022). Panels (top to bottom) illustrate changes to the cross-section (left) and planform (right) at agg/deg line 1549 (subreach BDA3; ~7 mi downstream of US HWY380) between 1962 and 2012.

# 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 surveyed at fixed transects known as aggradation/degradation (agg/deg) lines – cross-sections of the channel and floodplain that are spaced approximately every 500 ft along the length of the Middle Rio Grande (Posner, 2017). Surveys have occurred approximately every 10 years (1962, 1972, 1992, 2002, 2012; Figure 5). 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 (Bui, 2016; Posner, 2017). 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 of the Rio Grande Silvery Minnow.

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 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, $\delta$ (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 San Acacia 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.

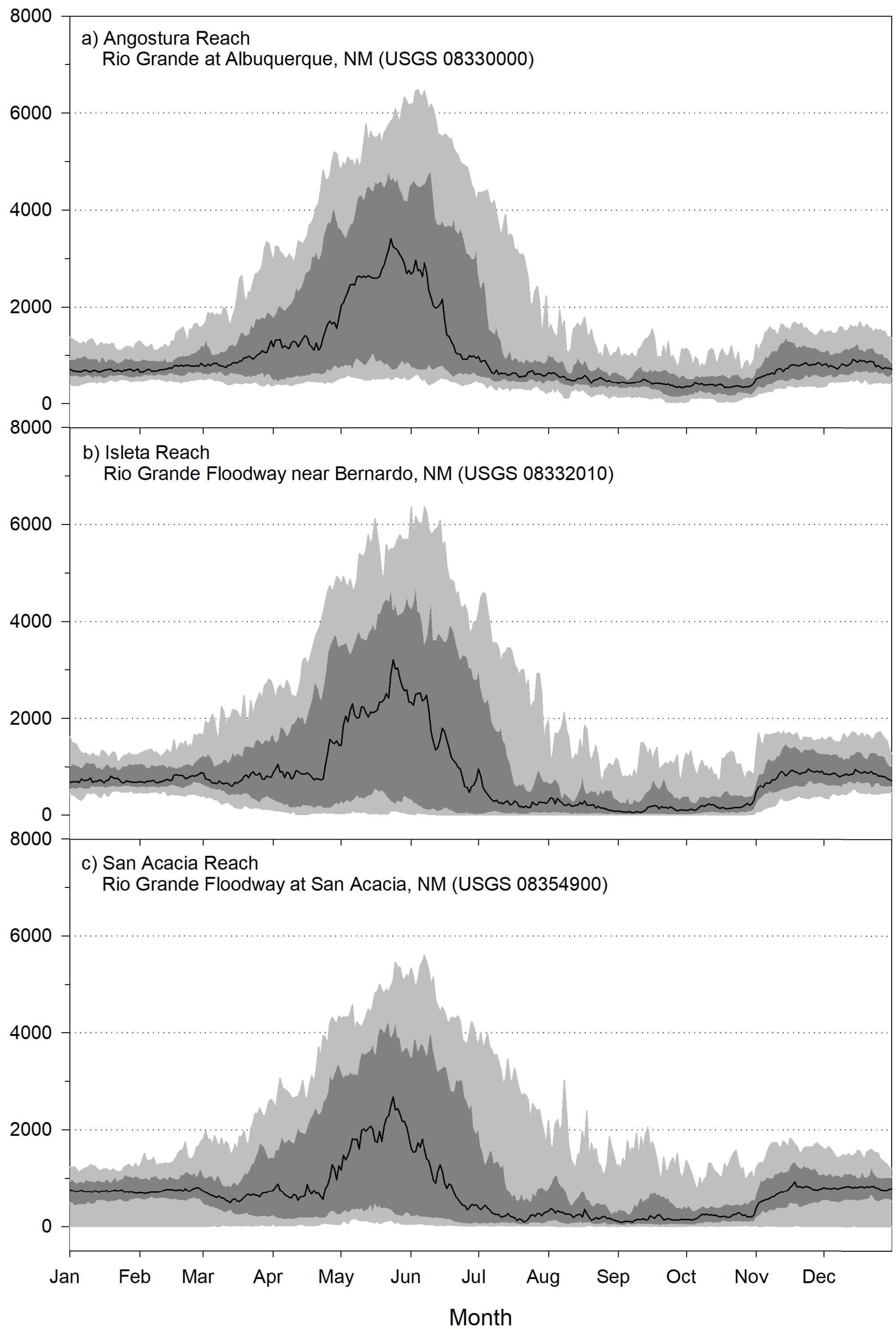



Figure 17. Representative annual hydrographs for Middle Rio Grande Reaches: a) Angostura, b) Isleta, and c) San Acacia. Black line shows median discharge ($Q_{50}$), dark gray shows interquartile range ($Q_{25}$ to $Q_{75}$), and light gray shows 10th and 90th percentiles ($Q_{10}$; $Q_{90}$) 1975–2020.

## Methods

### Subreach Delineation

Subreaches were delineated in the study area based on physical boundaries and geomorphic characteristics (Figure 3, Table 1; Doidge et al., 2020; Beckwith and Julien 2020; Schied et al., 2022; Sperry et al., 2022). 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 San Acacia Reach are presented in respective reach reports (Doidge et al., 2020; Beckwith and Julien, 2020; Schied et al., 2022; Sperry et al., 2022).

### 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 (1962–2012)

The estimation of bankfull discharges for each of the four subreach units (i.e., San Acacia, Escondida, Bosque del Apache, and Elephant Butte) was required due to complex channel morphology and computational limitations of the hydraulic modeling software, HEC-RAS, to accurately model such conditions. To improve the accuracy of one-dimensional hydraulic models of perched channels (i.e., channel bed elevation higher than floodplain [perched] or bank elevations higher than floodplain [semi-perched]), it is imperative to estimate bankfull discharges, as bankstation levees must be modeled when flows are less than bankfull. HEC-RAS distributes water from the bottom up, starting at the lowest elevations. This is problematic for locations of the river where the channel bed or banks are above the floodplain because HEC-RAS will inaccurately distribute water onto the floodplains at discharges below bankfull. To mitigate this computational inaccuracy, the method described by Baird and Holste (2020) was used to estimate bankfull discharge and assign top of bank points (i.e., computational levees) to prevent the lateral distribution of water before the banks overtopped. First, computational levees were placed at the locations that best represented the top of bank points on both sides of the channel. Thereby, water was restricted to the main channel (between the top of bank points) until water surface elevation exceeded the top of bank points. Left and right levee freeboard values, the difference between water surface elevations and the computational levee elevations, were used to determine the occurrence of overtopping by cross-section — negative values indicated overtopping. Due to spatial variation in cross-sections within reaches, a threshold value was selected using the percentage of cross-section overtopping within each reach (e.g., Escondida [E1–E5]). A threshold of 25% was deemed to be a reasonable approximation of bankfull discharge based on guidance from USBR. At discharges exceeding the 25% threshold of overtopped cross-sections, the top of bank points were removed from the model, thereby allowing water to inundate the modeled floodplain. For discharges less than the estimated bankfull discharge for the given reach, computational levees remained in place to minimize the inaccurate lateral distribution of water.

*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 and Julien, 2019). 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 [5 subdivisions] and floodplain [20 subdivisions; east and west]). Because floodplains can provide spawning and nursery habitats for the Rio Grande Silvery Minnow and contain more topographic variability than the main channel, 20 width-slices were assigned in each floodplain and 5 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 (terrain). The terrain is generated using LiDAR, which provides known ground surface elevations that represent the topography of the riparian area. 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 terrain (digital elevation model) for the San Acacia Reach. Using the steady flow data output from HEC-RAS, RAS Mapper distributed depth and velocity values over the San Acacia Reach terrain for several specified discharges (1,500, 3,000, and 5,000 cfs). To calculate the area of hydraulically suitable habitat by subreach and life-stage, the generated RAS Mapper results were imported to ArcMap 10.6 (ESRI, 2018). A model was developed using the ModelBuilder feature of ArcMap. 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.

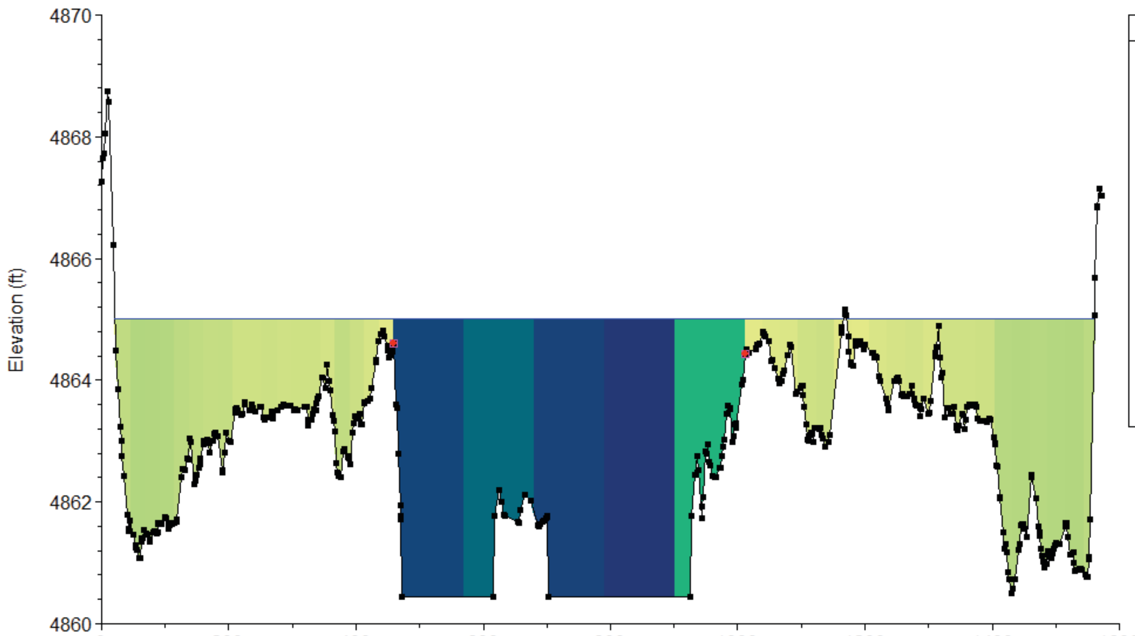



Figure 18. HEC-RAS graphic output showing the lateral flow distribution at a cross-section (width-slices). Width-slices were distributed with 5 slices in the main channel and 20 slices on each floodplain (n = 45 total slices per cross-section).

### *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 (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., SA1–SA4, E1–E5), 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.

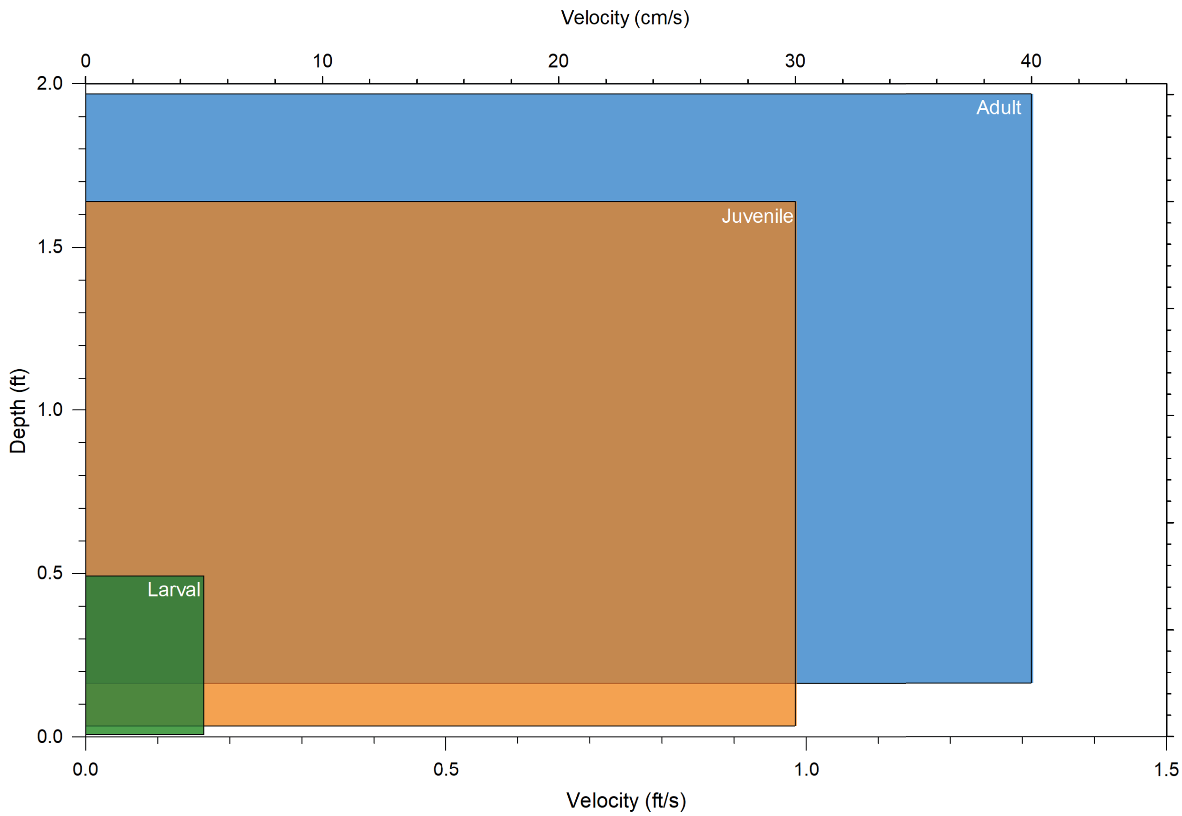



Figure 19. Hydraulically suitable habitat criteria (water velocity and depth) for the Rio Grande Silvery Minnow life-stages: adult, juvenile, and larval. Modified from Mortensen et al., 2019.

*Time Integrated Habitat Metrics (1993–2021)*

Habitat availability curves were integrated over time to calculate Time Integrated Habitat Metrics (TIHMs) for the San Acacia (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 (e.g., San Acacia Reach). Inputs required to calculate the TIHMs are daily discharge and flow-habitat curves for each decadal survey. Discharge data for the San Acacia Reach were obtained from the following USGS gaging stations: the Rio Grande floodway at San Acacia, NM (USGS 08354900; 1993–2021) the Rio Grande at bridge near Escondida, NM (USGS 08355050; 2006–2021), the Rio Grande above US HWY 380 near San Antonio, NM (USGS 08355490; 2006–2021), and the Rio Grande floodway at San Marcial, NM (USGS 08358400; 1993–2021). Habitat availability time series were calculated for each subreach using the nearest gaging station as hydrologic data availability permitted. 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 $\Delta T$ = 1 day. Units of the habitat metrics are: TIHM ($ft^2$ day/mi) = $H_t$ ($ft^2$/mi) •$\Delta T$ (day).

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 habitat metrics 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 08354900 (Rio Grande floodway at San Acacia, 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 across subreaches.

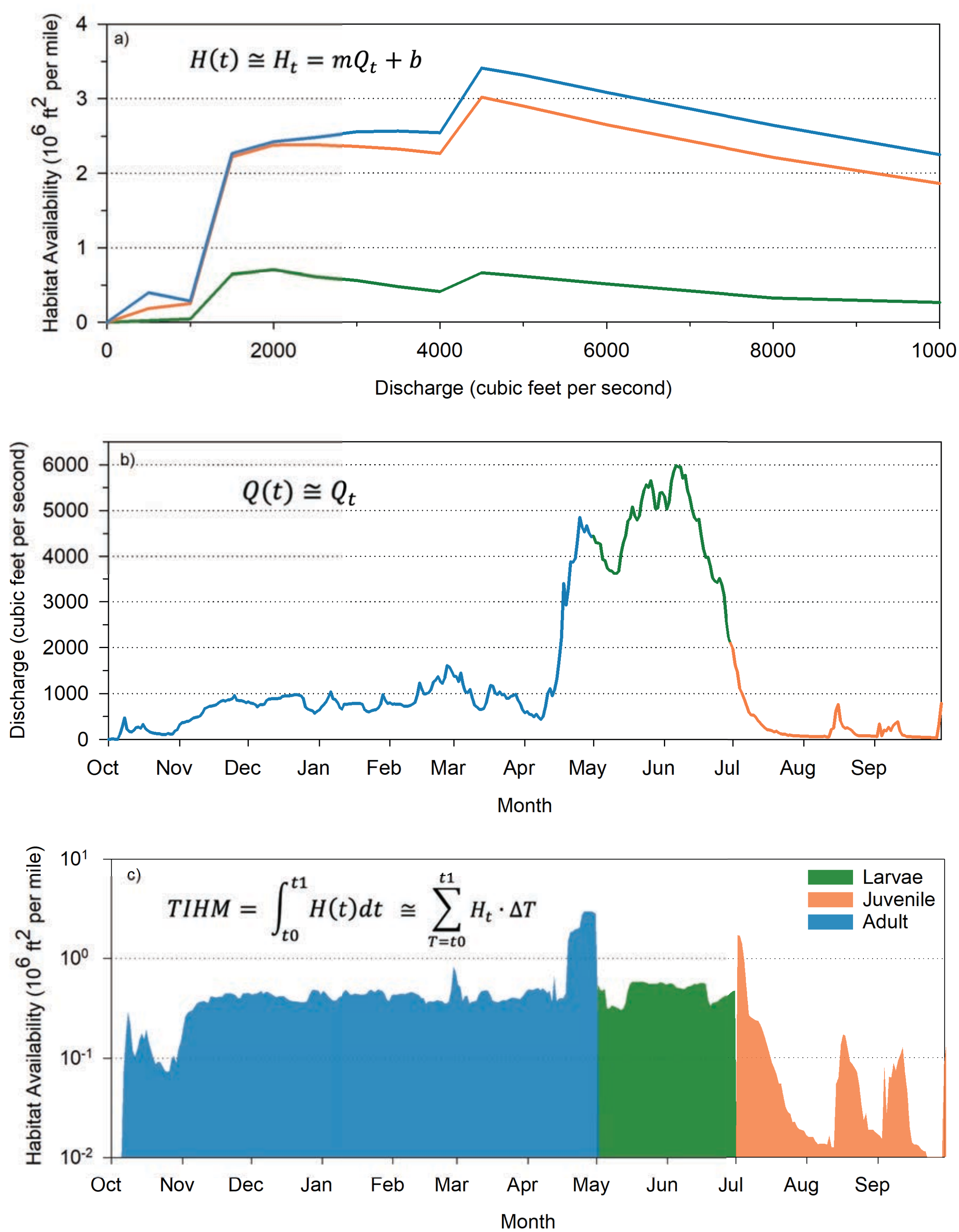



Figure 20. Example flow-habitat curves (a), hydrograph (b), and habitat availability time series (c) for the Isleta Reach during WY 2005. Shaded areas in the bottom panel represent TIHMs by life-stage (i.e., area under the curve, integral). Note log-scale in bottom panel.

*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 San Acacia 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 dynamics of the Rio Grande Silvery Minnow were evaluated at the reach-scale (i.e., San Acacia Diversion Dam to San Marcial, NM). 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, Juvenile: July–September, and Adult: October–April); the same periods that were used to calculate TIHMs. Additional flow metrics were included to characterize spring runoff and low flow conditions during life-stage periods. 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 population of the Rio Grande Silvery Minnow can indicate underlying ecological processes that drive population responses over time.

*Statistical Analyses*

Statistical analyses were performed using robust techniques suited for long-term ecological monitoring studies, such as the Rio Grande Silvery Minnow Population Monitoring Program. 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. 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 2021). 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). We implemented this ecological modeling approach to quantitatively assess the effects of environmental variables on long-term trends in the occurrence and density of the Rio Grande Silvery Minnow (Dudley et al. 2022). 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 (Appendix D). Numerical optimization of the models provided four estimates ($\delta$ = estimated occurrence probability, $\mu$ = estimated lognormal density, $\sigma$ = standard deviation of the estimated lognormal density, and $E(x)$ = estimated density) for each year (i.e., based on the site-specific sampling data). Values of $E(x)$ could not be computed, however, when only a single nonzero value was recorded (i.e., precluding mixture-model estimation of $\sigma$). Naïve density estimates (i.e., unmodeled), calculated using the method of moments (Zar 2010), were also added as a reference to applicable figures.

Long-term ecological relationships were evaluated using generalized linear models. Generalized linear models were based on environmental covariates (i.e., independent variables) and population parameter estimates ($\delta$, $\mu$, and $\sigma$ [i.e., dependent variables]), where a logit link was used for $\delta$, an identity link for $\mu$, and a log link for $\sigma$. The logit link maintains delta on a 0–1 scale, the identity link maintains the mean of the lognormal distribution between –infinity and +infinity, and the log link maintains sigma (SD of lognormal distribution) 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 ($\delta$[Year] $\mu$[Year]) represents the upper limit on the amount of explainable variation and the null model ($\delta$[.] $\mu$[.]) represents the lower limit of that variation. Additionally, all nested environmental covariates (e.g., spring and summer flows) varied across Year and were assessed individually as to 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 TIHM estimates were calculated for larval habitat (May to June 2021), juvenile habitat (July to September 2021), and adult habitat (October 2020 to April 2021); month ranges coincided with the highest prevalence of each life phase. Habitat metrics 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 direct 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 $\delta$, there is no over-dispersion or extra-binomial variation, and for $\mu$, no extra variation provided beyond the constant $\sigma$ model. Random-effects models (*R*) were also considered for $\delta$ and $\mu$ 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] for finite sample sizes) were generated to assess the relative fit of data to various mixture models across all sampling years. 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 $\delta$ and $\mu$ 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.

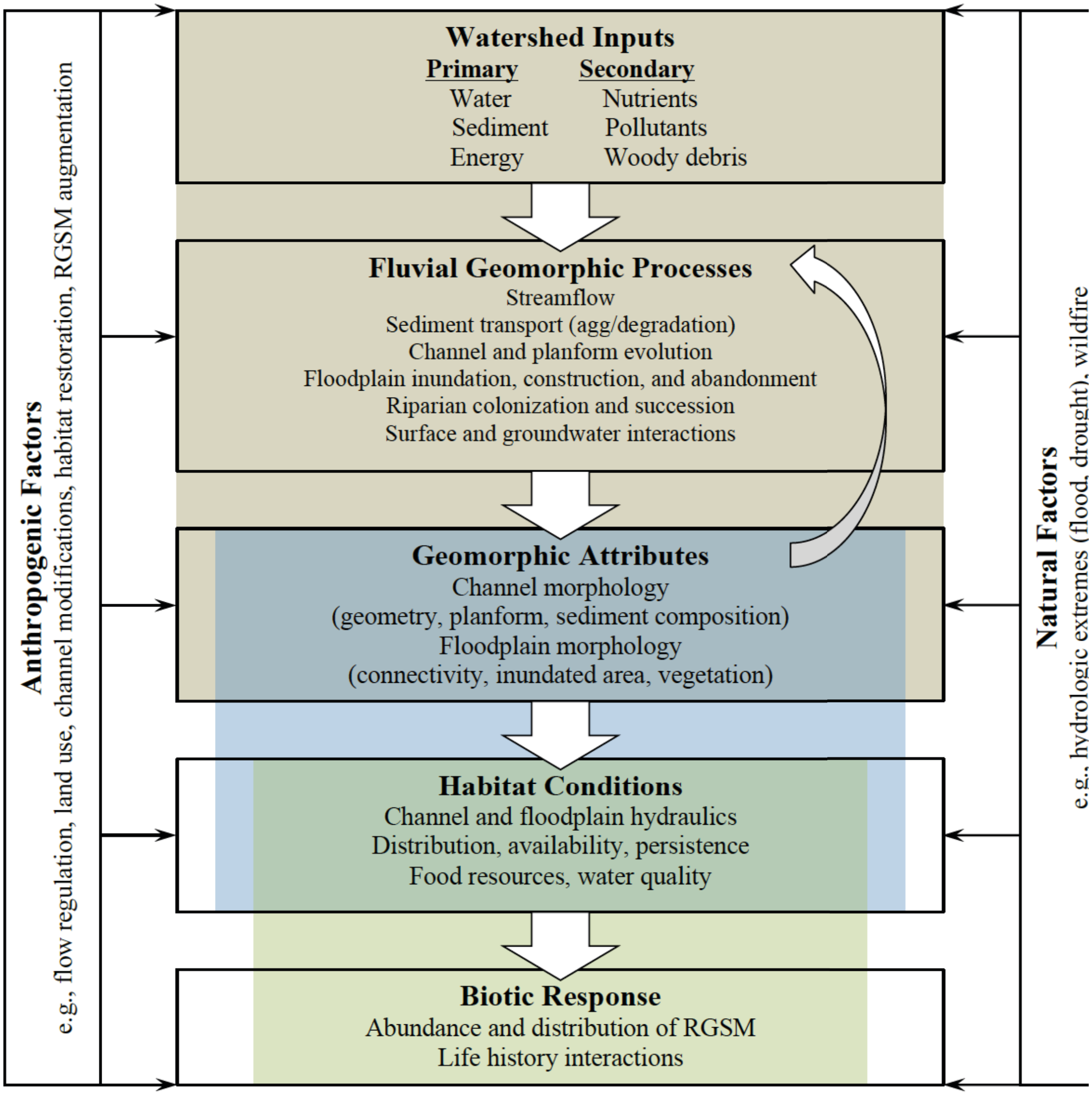



Figure 21. Simplified conceptual model of the linkages between watershed inputs, fluvial geomorphic processes and attributes, habitat conditions, and the biota of the Middle Rio Grande ecosystem. Colored boxes represent nested sub-models: Fig. B (brown), Fig. C (blue), and Fig. D (green). Modified from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

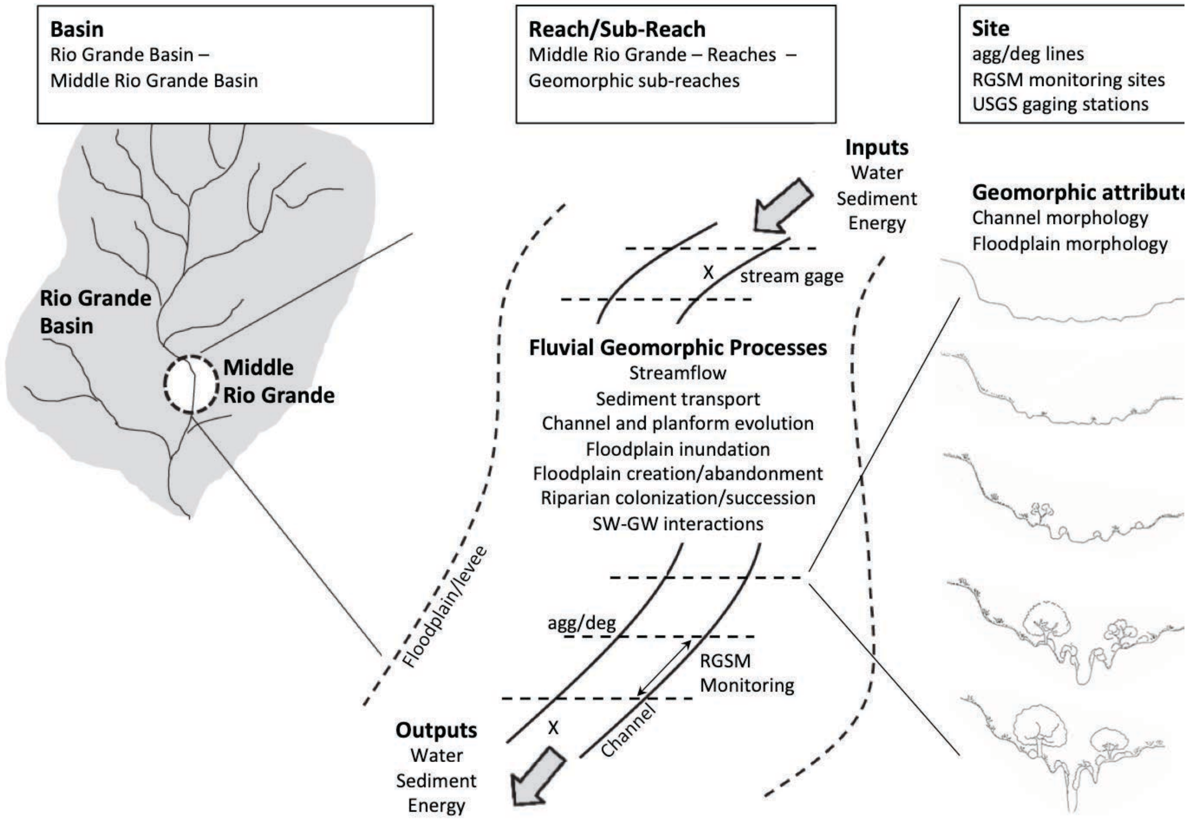



Figure 22. Delineation of spatial units used in this study, including the interactions among watershed inputs, fluvial geomorphic processes, and geomorphic attributes, and long-term systematic data collection efforts in the Middle Rio Grande (e.g., agg/deg lines, RGSM monitoring sites, stream gages).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

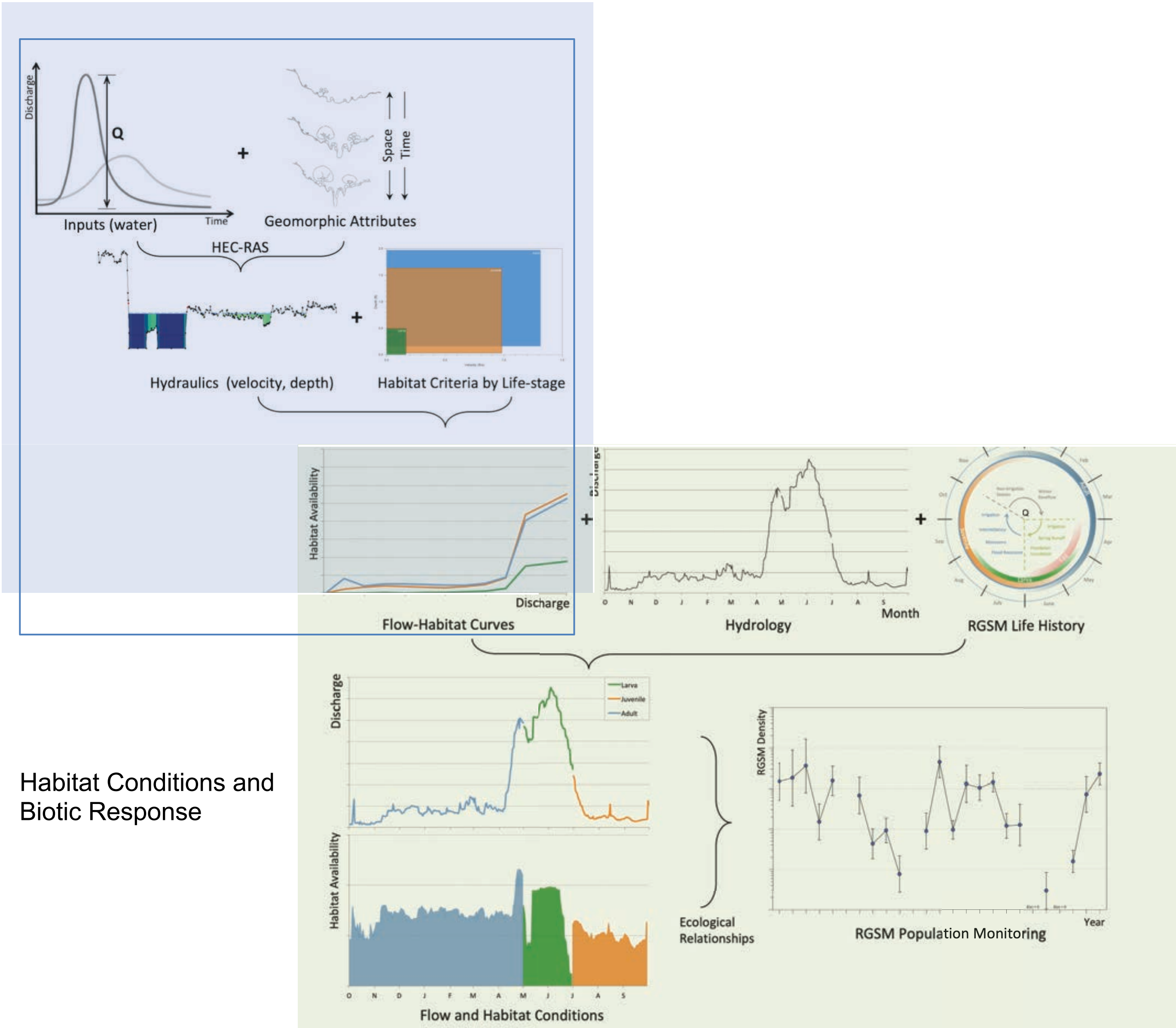



Figure 23. Characterization and assessment of geomorphic attributes, hydraulics, and physical habitat conditions through time including their interaction with environmental factors and the life history of the RGSM. Colored boxes (blue, green) correspond to Figure 21.

# RESULTS

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

## *Reach and Habitat Analyses – San Acacia Reach*

### *Bankfull Discharge (1962–2012)*

Reach-averaged bankfull discharges were estimated for the San Acacia (SA1–SA4), Escondida (E1–E5), Bosque del Apache (BDA1–BDA5), and Elephant Butte (EB1–EB5) subreaches individually (Figure 24). In subreaches SA1–SA4, the lowest bankfull discharges occurred in 1962 and 1972 at approximately 4,000 cfs, with minimal change between these survey years. Bankfull discharges increased dramatically between 1972 and 1992 to greater than 10,000 cfs in this subreach. Bankfull discharge remained elevated until 2012 with a decrease to about 7,000 cfs. In subreaches E1–E5, bankfull discharge decreased from 1962 to 1972 from about 4,000 cfs to about 2,500 cfs. Bankfull discharge increased from 1972 to 1992 to approximately 4,000 cfs, where it appears to have remained relatively constant through 2012. Subreaches BDA1–BDA5 showed the lowest magnitude of change in bankfull discharge during the study period. Bankfull discharges in this subreach remained between about 1,500 cfs (1962, 1992, 2002) and 2,000 cfs (1972, 2012). Subreaches EB1–EB5 showed the most variable change in bankfull discharge during the study period. Bankfull discharges in this subreach was about 2,500 cfs 1962–1972, with a decrease to about 1,000–1,500 cfs 1992–2002, and a sharp increase to about 4,500 cfs in 2012. Estimated bankfull discharges were important for improving modeling accuracy of floodplain inundation and habitat availability at low flows (i.e., discharges below bankfull); these values were used to approximate the threshold at which the floodplain becomes connected to the main channel.

### *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 25). 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 25, Figure A-1).

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, however, juvenile habitat availability was typically lower than adult habitat availability across discharges. Larval habitat availability was lowest in magnitude and generally displayed similar trends to the juvenile and adult life-stages. All flow-habitat curves showed sharp increases in habitat availability with increasing discharge corresponding to the respective bankfull discharge of the subreach and survey year, except for the SA subreach during 1992–2012, due to large increases in bankfull discharge that occurred prior to and during this period.

Habitat availability varied temporally and spatially during the study period (Figure 25). Notably, the SA subreach showed dramatic decreases to habitat availability across the range of discharges analyzed for survey years 1992–2012 related to channel narrowing and incision. In contrast, downstream subreaches maintained relatively high habitat availability across discharges throughout the study period. Subreaches E1–E5 showed a marginal shift associated with increasing bankfull discharge 1972–1992, however, the magnitude of habitat availability remained high 1992–2012 relative to 1962–1972 at discharges greater than bankfull (>3,000–4,000 cfs). Subreaches BDA1–BDA5 showed the lowest magnitude of change during the study period. In this subreach, the magnitude of habitat availability remained high with marginal changes to bankfull discharge across all survey years. Subreaches EB1–EB5 showed somewhat variable trends during the study period with increased habitat availability at lower flows 1992–2002 relative to 1962–1972 and a notable decrease in habitat availability at low to moderate flows (<4,500 cfs) during 2012. Defining trends observed in this analysis include decreased habitat availability over time in the SA subreach, maintenance of habitat availability over time in the Escondida and BDA subreaches, and variable trends in habitat availability over time for the EB subreach.

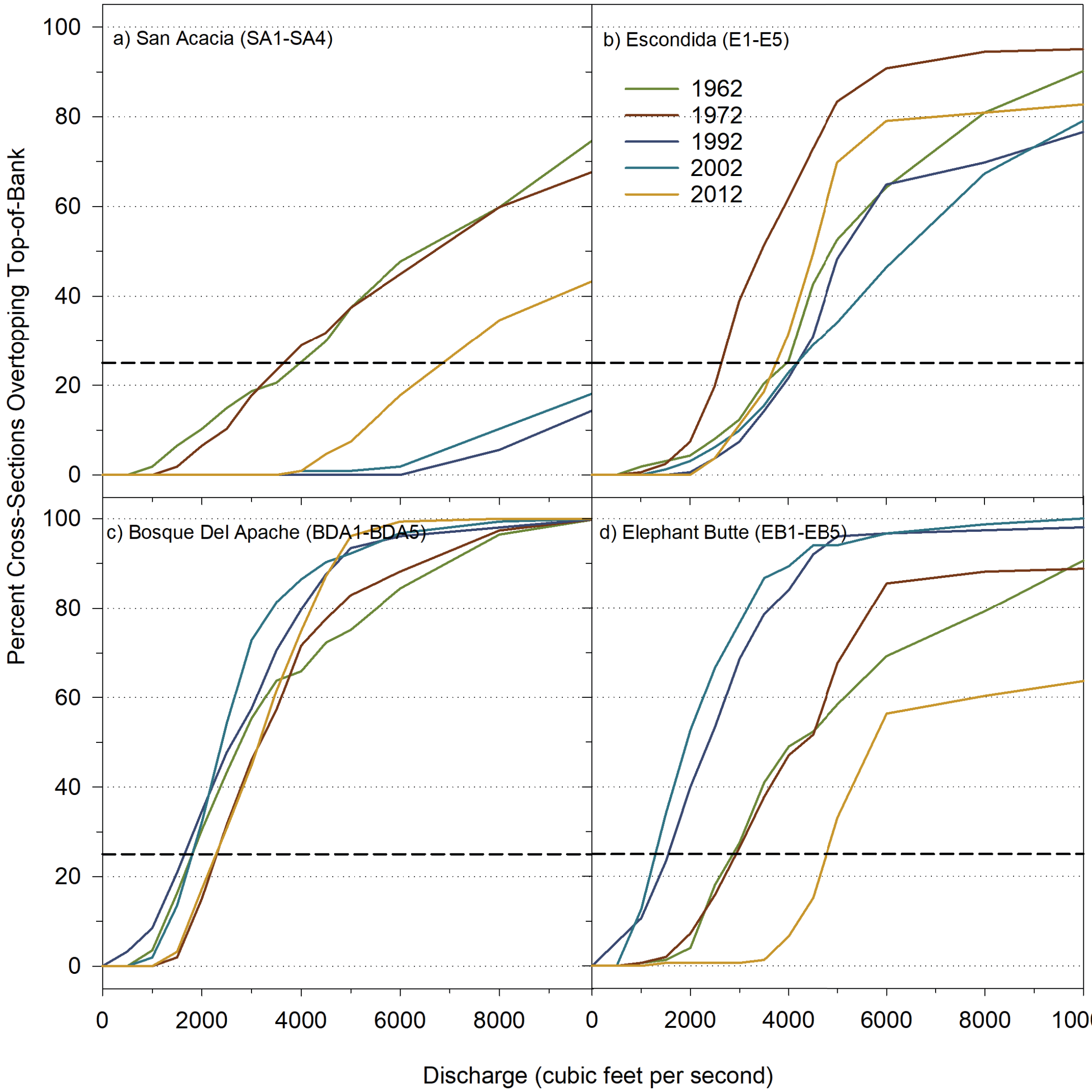



Figure 24. Estimated bankfull discharge for San Acacia (a), Escondida (b), Bosque del Apache (c), and Elephant Butte (d) subreaches for available channel geometry datasets (1962–2012). Bankfull discharge was estimated to occur when the percentage of overtopped cross-sections exceeded 25% (dashed black line).

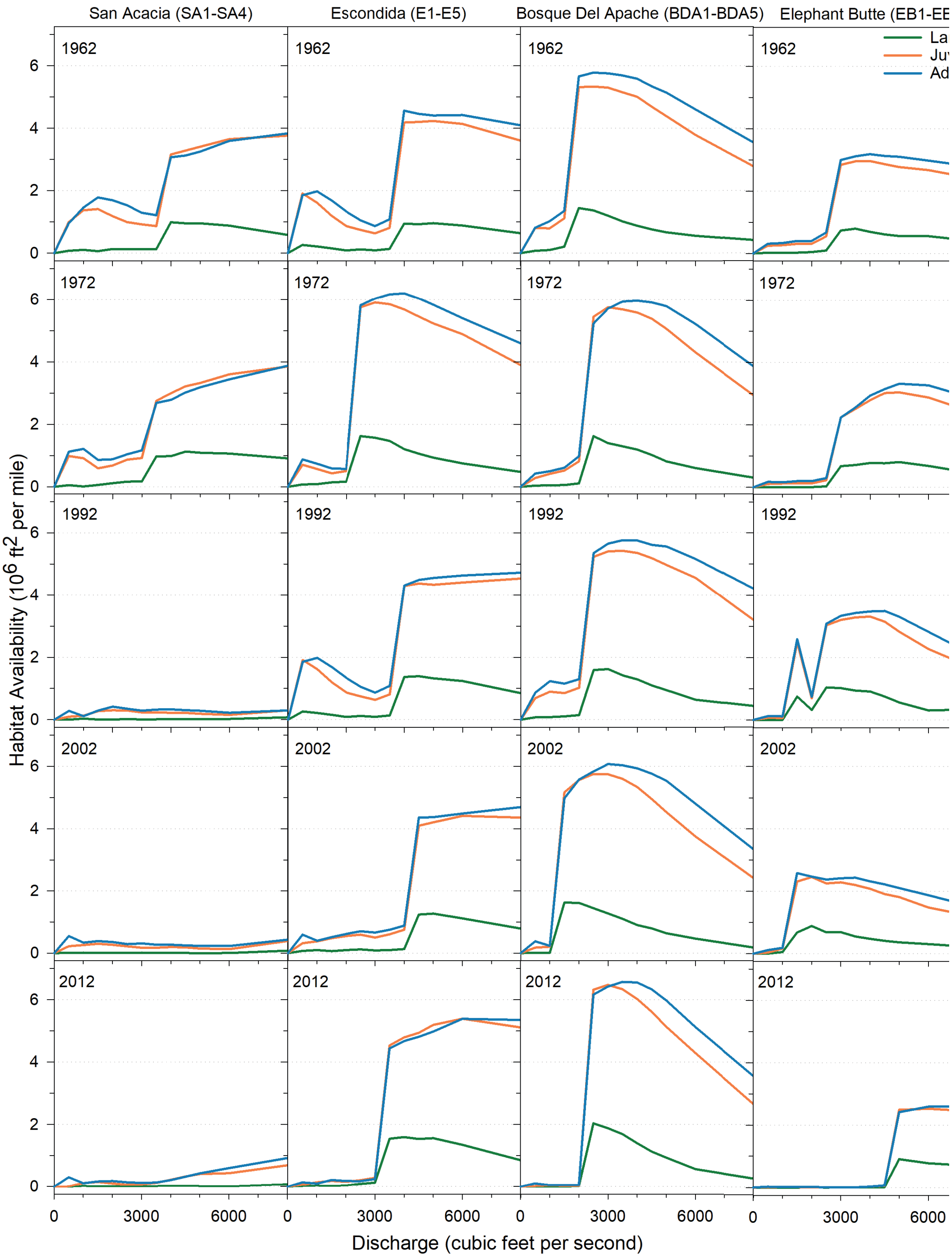



Figure 25. Flow-habitat curves for the San Acacia Reach. Columns left to right are subreaches. Curves are shown through time top to bottom (1962–2012). Line colors represent the primary life-stages of the Rio Grande Silvery Minnow. Habitat availability was normalized by reach length.

### *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 25), San Acacia Reach discharge values (Figure 26), and life-stage periods were the inputs used to calculate TIHMs. A total of 87 TIHMs were calculated for the study period (Table 4).

Discharge data were selected for each subreach where available. USGS gaging stations were located coincident with the upstream end of each subreach (Figure 26). Data were applied as follows: San Acacia subreach – USGS 08354900 (Rio Grande floodway at San Acacia, NM; 1993–2021), Escondida subreach – USGS 08354900 (1993–2005), USGS 08355050 (Rio Grande at bridge near Escondida, NM; 2006–2021), Bosque del Apache subreach – USGS 08355490 (Rio Grande floodway at San Marcial, NM; 1993–2005),USGS 08355490 (Rio Grande above US HWY380 near San Antonio, NM; 2006–2021), Elephant Butte subreach – USGS 08358400 (1993–2021). Hydrologic data in the San Acacia Reach accounted for spatial variability in hydrologic conditions 2005–2021.

Time Integrated Habitat Metrics varied through time and by life-stage period (Figure 27–28). Larval TIHMs ranged from 0.02–52.03 $10^6$ $ft^2$ day/mi, juvenile TIHMs ranged 0.12–113.38 $10^6$ $ft^2$ day/mi, and adult TIHMs ranged 20.23–201.65 $10^6$ $ft^2$ day/mi. Mean TIHMs for the study period were lowest for the larval life-stage (13.04 ± 15.24 $10^6$ $ft^2$ day/mi) and were followed by the juvenile life-stage (23.62 ± 28.67 $10^6$ $ft^2$ day/mi). Adult TIHMs were the highest on average (99.31 ± 60.74 $10^6$ $ft^2$ day/mi) and had the highest variance of the life-stage periods. Median TIHMs were highest for adults (98.28 $10^6$ $ft^2$ day/mi), followed by juveniles (8.20 $10^6$ $ft^2$ day/mi), and were lowest for larvae (5.38 $10^6$ $ft^2$ day/mi). Subreach contributions to habitat metrics were varied, reflecting spatial variation in hydrologic and geomorphic conditions over time. Escondida and Bosque del Apache subreaches generally contributed the most habitat over time and life-stage period. 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 were strongly influenced by the annual hydrograph, specifically seasonal flow conditions corresponding to each life-stage period (Figures 29). 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 larvae were also sensitive to estimated bankfull discharges, which acted as a threshold or step function for habitat availability during peak flows May–June. TIHMs for adults were less affected by discharge fluctuations due to the tendency for relatively stable conditions to occur October–April.

### *Effects of Geomorphic Changes on Habitat Metrics*

The effects of temporal geomorphic changes on habitat metrics (TIHMs) 1962–2012 were assessed using selected annual hydrographs (Figure 30). Discharge data inputs were held constant for the reach (USGS 08354900 Rio Grande floodway at San Acacia, NM). Due to the strong influence of annual hydrographs on the TIHMs (Table 4, Figure 29), hydrologic conditions were isolated to assess the relative influence of geomorphic changes on the habitat metrics.

Overall, Time Integrated Habitat Metrics decreased between 1962 and 2012 for each of the flow scenarios investigated except for the high flow scenarios specific to the larval life-stage (Table 5; Figures 31–33. 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 moderate and high flow scenarios showed relatively constant values during the study period in contrast to juvenile and adult life-stages. Between 1962 and 2012, juvenile and adult TIHMs showed relatively consistent decreases across flow scenarios – juvenile TIHMs on average decreased by 88–98% and adult TIHMs on average decreased by 77–92% across flow scenarios during this period. The larval TIHMs, however, showed varied trends across flow scenarios between 1962 and 2012. The highest relative decreases in larval TIHMs occurred for the low flow scenarios (95% on average), which was followed by the moderate flow scenarios (63% on average). Notably, the high flow scenarios showed a slight increase in larval habitat availability as measured by the TIHMs (8% on average). Subreach contributions to TIHMs varied by life-stage and flow scenario. Changes to subreach contributions were most pronounced for the larval life-stage. Subreach contributions tended to be dominated by E and BDA subreaches whereas SA and EB subreaches generally showed habitat loss over time.

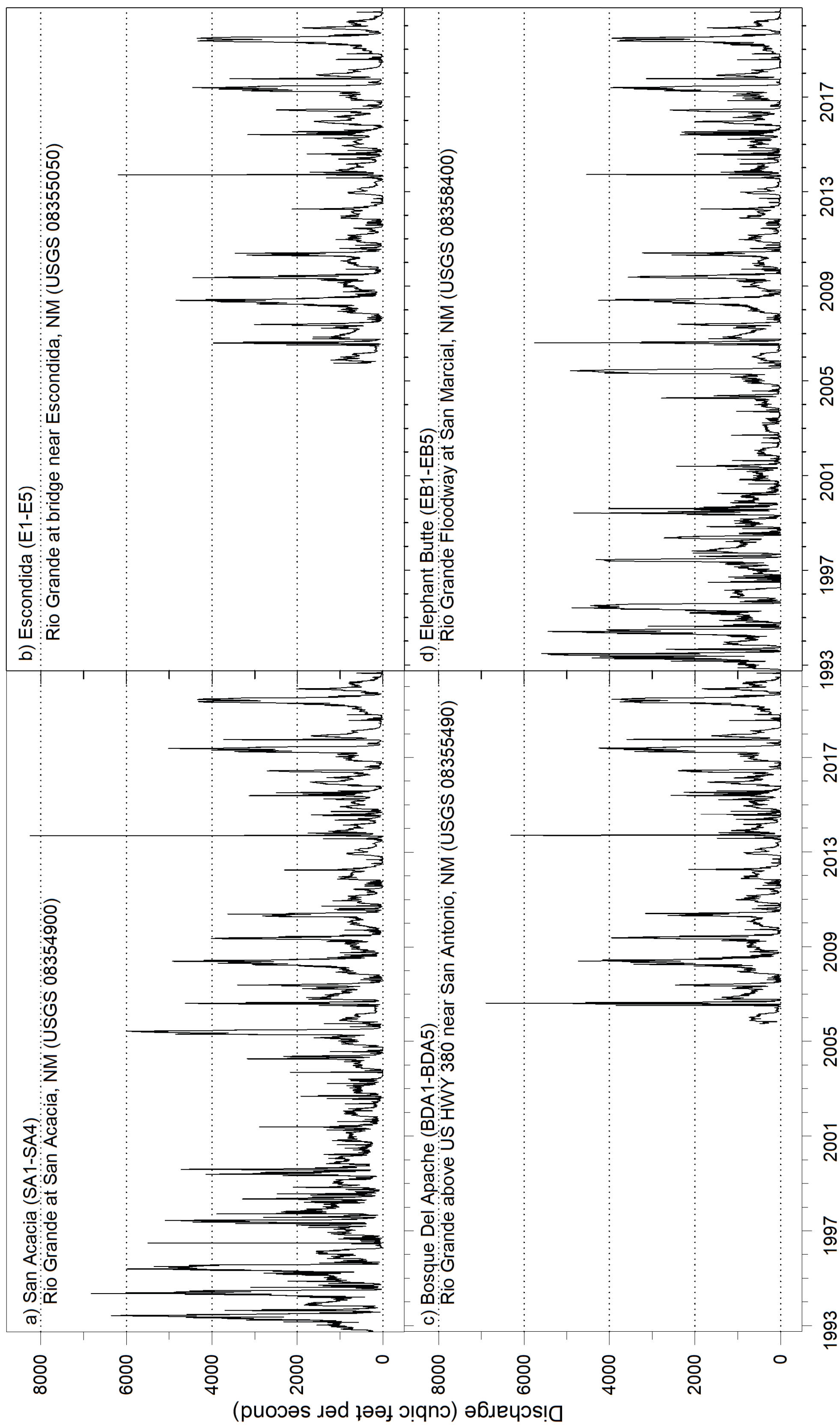
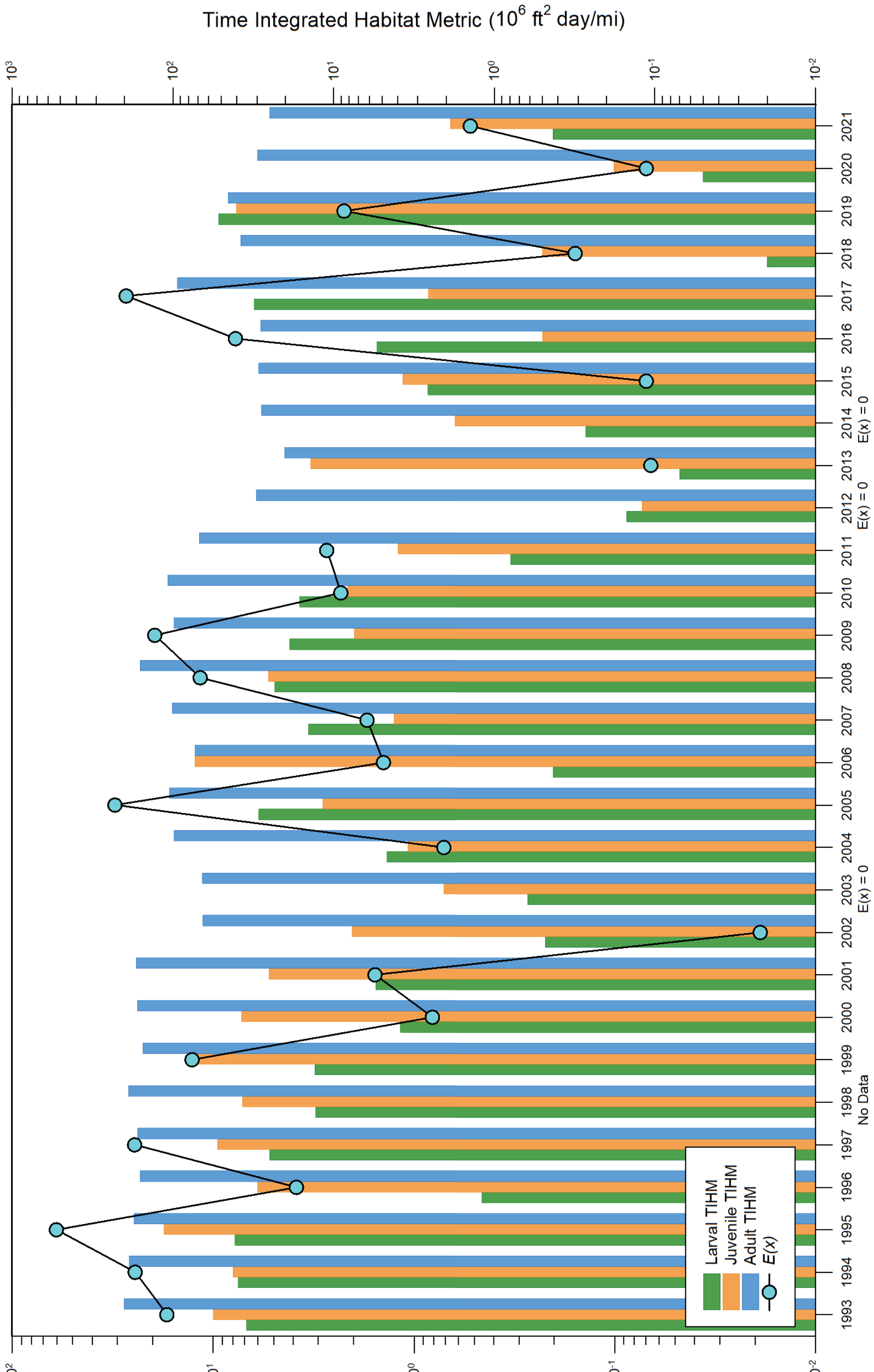

a) San Acacia (SA1-SA4)
Rio Grande at San Acacia, NM (USGS 08354900)
b) Escondida (E1-E5)
Rio Grande at bridge near Escondida, NM (USGS 08355050)
c) Bosque Del Apache (BDA1-BDA5)
Rio Grande above US HWY 380 near San Antonio, NM (USGS 08355490)
d) Elephant Butte (EB1-EB5)
Rio Grande Floodway at San Marcial, NM (USGS 08358400)
Discharge (cubic feet per second)
8000
6000
4000
2000
0
1993
1997
2001
2005
2009
2013
2017

Table 4. Time Integrated Habitat Metrics (TIHMs) by life-stage period for the San Acacia Reach 1993–2021.

| | TIHMs ($10^6$ ft$^2$ day per mile) | | |
|---|---|---|---|
| Water Year | Larval[1] | Juvenile[2] | Adult[3] |
| 1993 | 34.82 | 56.17 | 201.65 |
| 1994 | 39.25 | 42.27 | 186.91 |
| 1995 | 41.20 | 113.38 | 175.23 |
| 1996 | 1.19 | 29.78 | 159.99 |
| 1997 | 25.10 | 52.94 | 165.96 |
| 1998 | 12.96 | 37.06 | 188.82 |
| 1999 | 13.13 | 77.97 | 153.58 |
| 2000 | 3.85 | 37.36 | 167.03 |
| 2001 | 5.49 | 25.31 | 169.21 |
| 2002 | 0.48 | 7.68 | 65.16 |
| 2003 | 0.62 | 2.07 | 65.64 |
| 2004 | 4.67 | 3.46 | 98.42 |
| 2005 | 29.25 | 11.67 | 105.44 |
| 2006 | 0.43 | 73.21 | 72.83 |
| 2007 | 14.37 | 4.23 | 101.14 |
| 2008 | 23.27 | 25.50 | 160.02 |
| 2009 | 18.79 | 7.43 | 98.28 |
| 2010 | 16.28 | 8.20 | 107.71 |
| 2011 | 0.79 | 3.97 | 68.51 |
| 2012 | 0.15 | 0.12 | 30.32 |
| 2013 | 0.07 | 13.89 | 20.23 |
| 2014 | 0.27 | 1.76 | 28.11 |
| 2015 | 2.59 | 3.72 | 29.28 |
| 2016 | 5.38 | 0.50 | 28.43 |
| 2017 | 31.25 | 2.57 | 94.05 |
| 2018 | 0.02 | 0.50 | 37.80 |
| 2019 | 52.03 | 40.14 | 45.29 |
| 2020 | 0.05 | 0.18 | 29.77 |
| 2021 | 0.43 | 1.88 | 25.06 |

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


Time Integrated Habitat Metric ($10^6$ ft$^2$ day/mi)
$10^3$
$10^2$
$10^1$
$10^0$
$10^{-1}$
$10^{-2}$
2021
2020
2019
2018
2017
2016
2015
2014
E(x) = 0
2013
2012
E(x) = 0
2011
2010
2009
2008
2007
2006
2005
2004
2003
E(x) = 0
2002
2001
2000
1999
1998
No Data
1997
1996
1995
1994
1993
Larval TIHM
Juvenile TIHM
Adult TIHM
E(x)
$10^2$
$10^1$
$10^0$
$10^{-1}$
$10^{-2}$

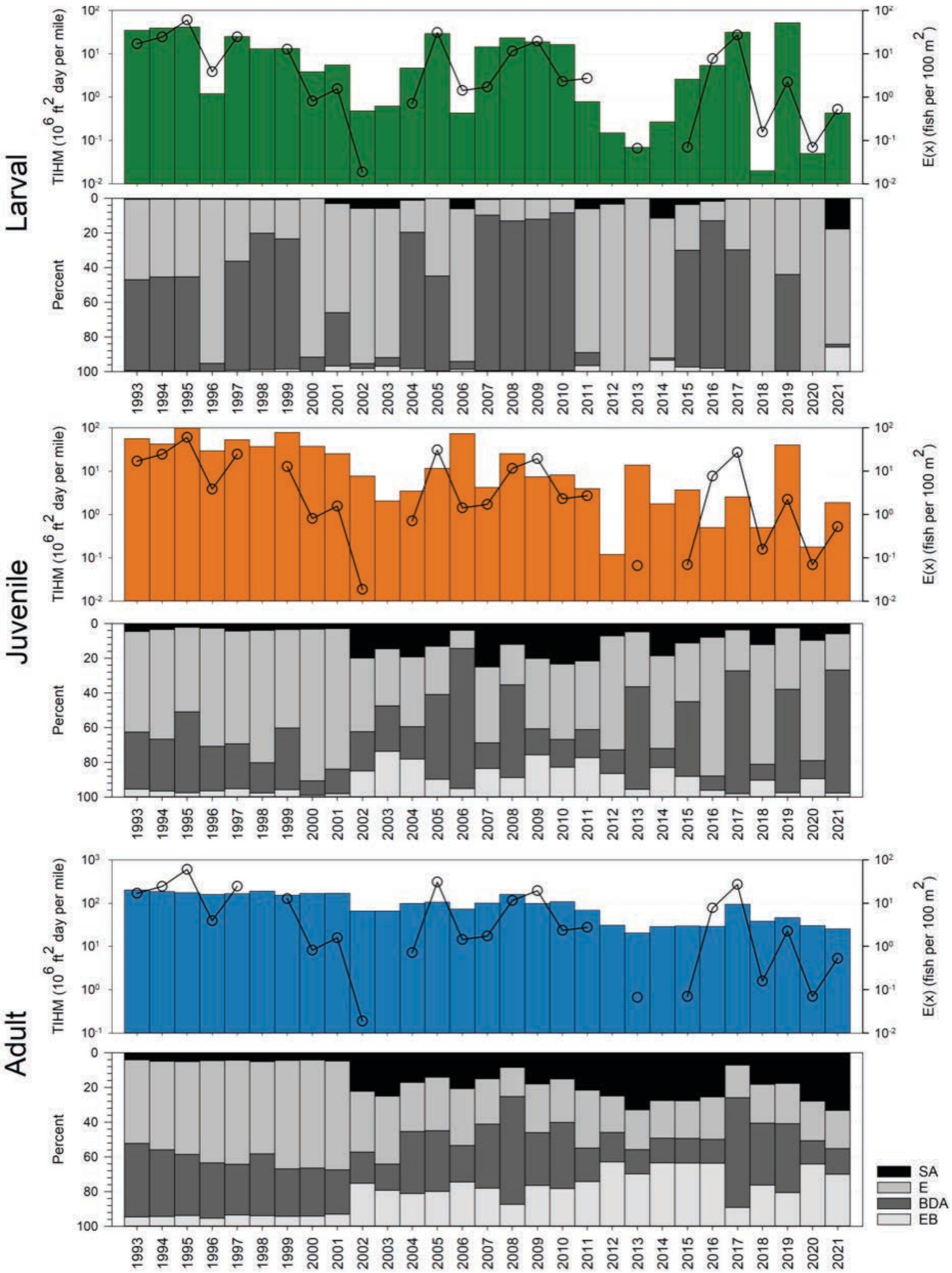
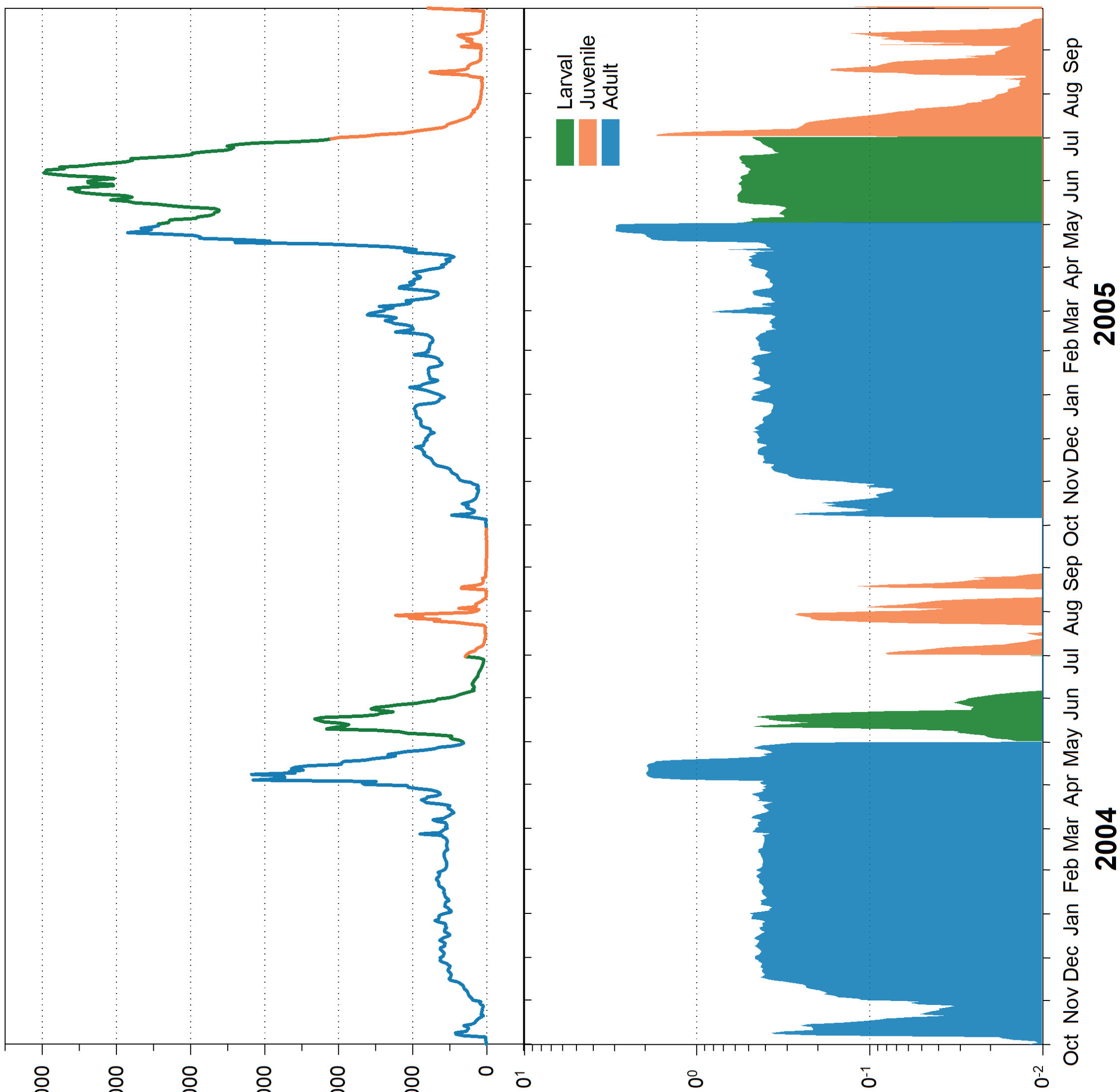



Figure 28. Annual Time Integrated Habitat Metrics (TIHMs; colored bars), by life-stages of the Rio Grande Silvery Minnow, during the study period (1993–2021), annual estimated densities of the Rio Grande Silvery Minnow in October (*E(x)*; black circles and lines), and percentage contribution by subreach (stacked bars). Note the log scale of the Y-axis


Larval
Juvenile
Adult
Oct Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep Oct Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep
2004
2005
0
$10^{1}$
$10^{0}$
$10^{-1}$
$10^{-2}$

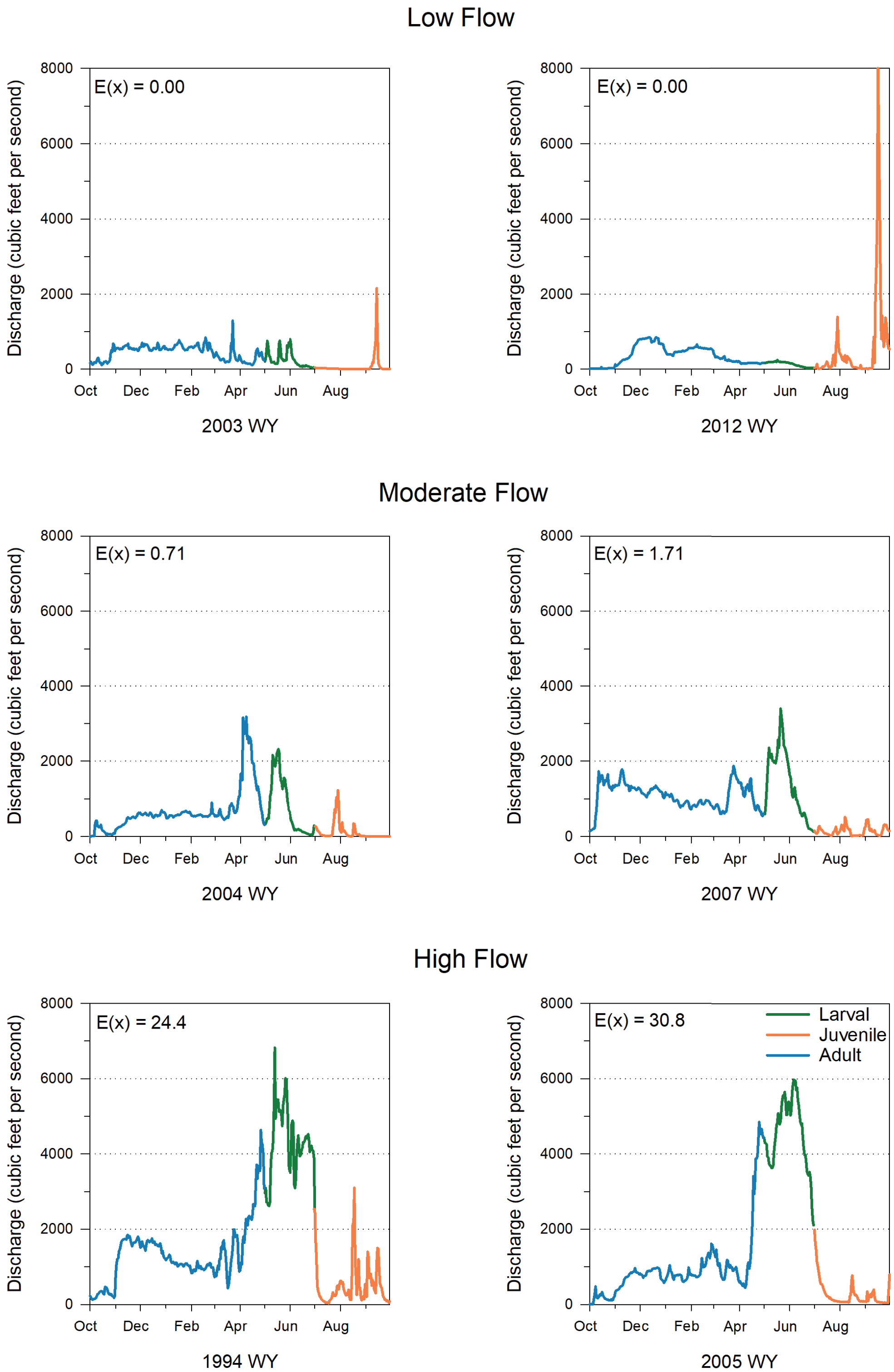



Figure 30. Selected annual hydrographs for assessment of geomorphic changes on habitat metrics. Hydrographs were selected to represent three flow scenarios: low, moderate, and high. Discharge data from USGS 08354900 Rio Grande floodway at San Acacia, NM. Estimated densities, E(x), are in units of number of fish per 100 $m^2$.

Table 5. Time Integrated Habitat Metrics for the San Acacia Reach calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes on habitat availability over time.

| | | | Time Integrated Habitat Metrics ($10^6$ $ft^2$ day/mi) | | |
|---|---|---|---|---|---|
| Water Year | Flow Scenario | Survey Year | Larval | Juvenile | Adult |
| 2003 | low | 1962 | 3.01 | 7.76 | 169.21 |
| | | 1972 | 1.14 | 4.20 | 102.34 |
| | | 1992 | 2.49 | 5.61 | 140.59 |
| | | 2002 | 0.70 | 3.57 | 65.43 |
| | | 2012 | 0.16 | 0.85 | 20.34 |
| 2012 | low | 1962 | 3.15 | 5.67 | 204.32 |
| | | 1972 | 1.20 | 2.77 | 116.08 |
| | | 1992 | 2.62 | 4.26 | 166.87 |
| | | 2002 | 0.68 | 1.07 | 76.68 |
| | | 2012 | 0.15 | 0.12 | 30.32 |
| 2004 | moderate | 1962 | 7.60 | 15.02 | 209.17 |
| | | 1972 | 4.42 | 7.26 | 143.91 |
| | | 1992 | 7.14 | 11.52 | 178.41 |
| | | 2002 | 12.36 | 4.16 | 112.76 |
| | | 2012 | 1.79 | 0.58 | 40.57 |
| 2007 | moderate | 1962 | 15.76 | 25.16 | 252.56 |
| | | 1972 | 14.39 | 12.13 | 122.48 |
| | | 1992 | 14.58 | 18.91 | 228.96 |
| | | 2002 | 21.29 | 4.74 | 209.64 |
| | | 2012 | 7.82 | 0.60 | 18.98 |
| 1994 | high | 1962 | 45.22 | 62.22 | 299.89 |
| | | 1972 | 59.08 | 35.49 | 181.72 |
| | | 1992 | 49.56 | 50.17 | 256.20 |
| | | 2002 | 33.22 | 31.81 | 278.97 |
| | | 2012 | 47.79 | 7.22 | 63.09 |
| 2005 | high | 1962 | 44.32 | 35.73 | 246.71 |
| | | 1972 | 56.71 | 16.21 | 167.10 |
| | | 1992 | 46.19 | 26.91 | 209.87 |
| | | 2002 | 33.51 | 13.52 | 126.93 |
| | | 2012 | 49.13 | 1.21 | 56.92 |

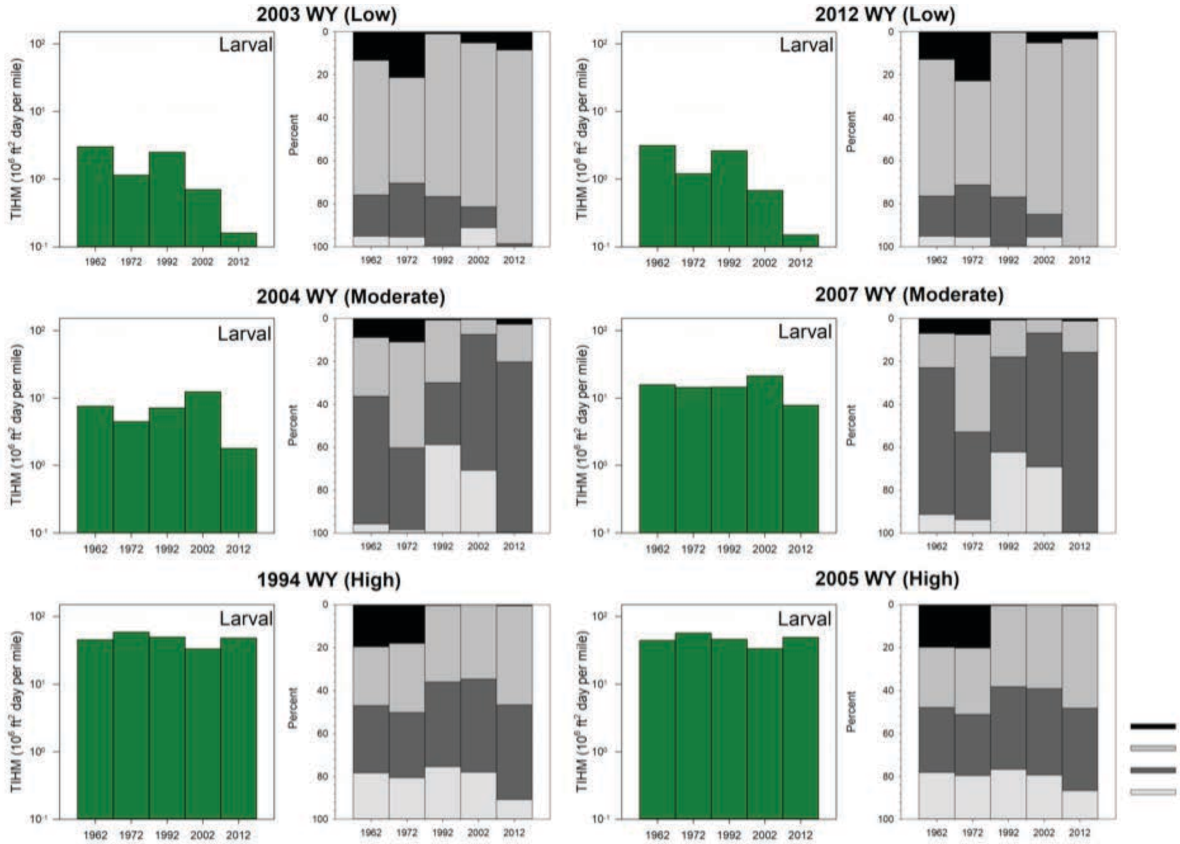



Figure 31. Time Integrated Habitat Metrics (TIHMs; colored bars) for the larval life-stage and percentage contribution by subreach (stacked bars) calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes over time.

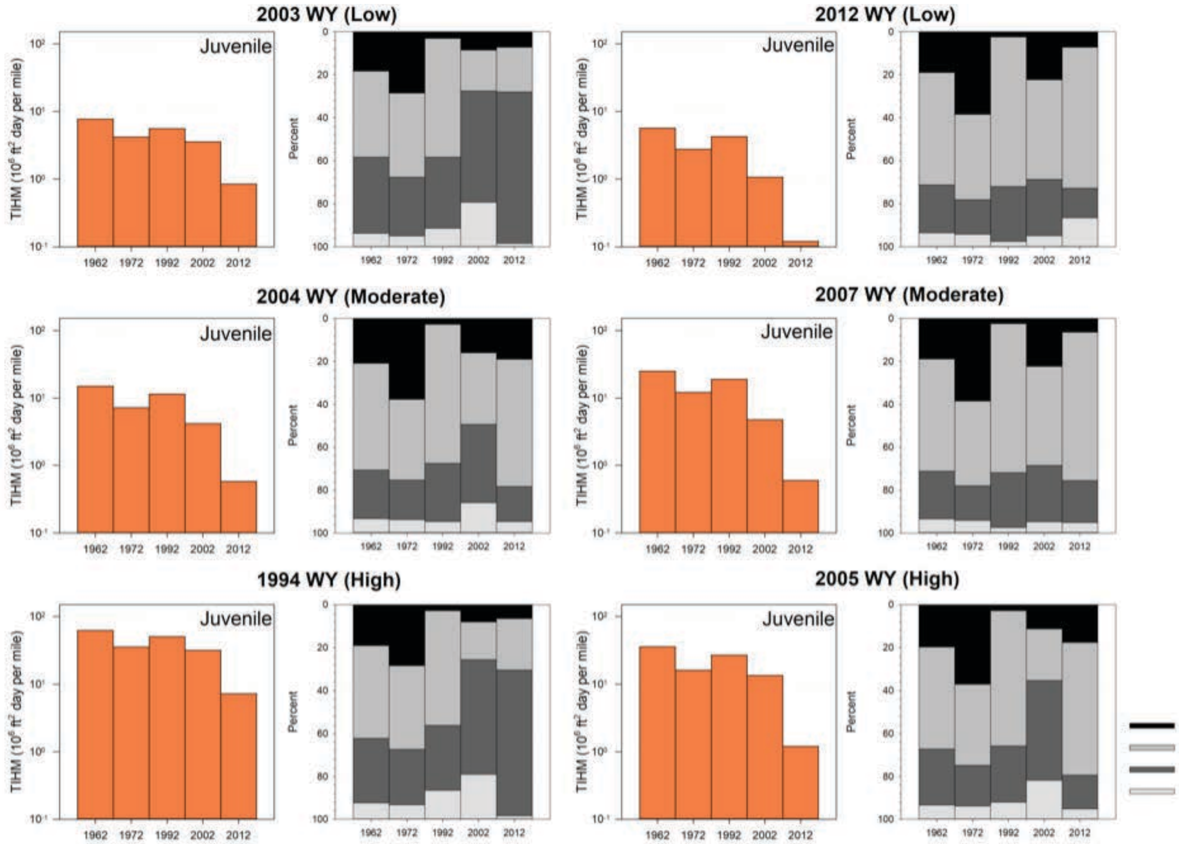



Figure 32. Time Integrated Habitat Metrics (TIHMs; colored bars) for the juvenile life-stage and percentage contribution by subreach (stacked bars) calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes over time.

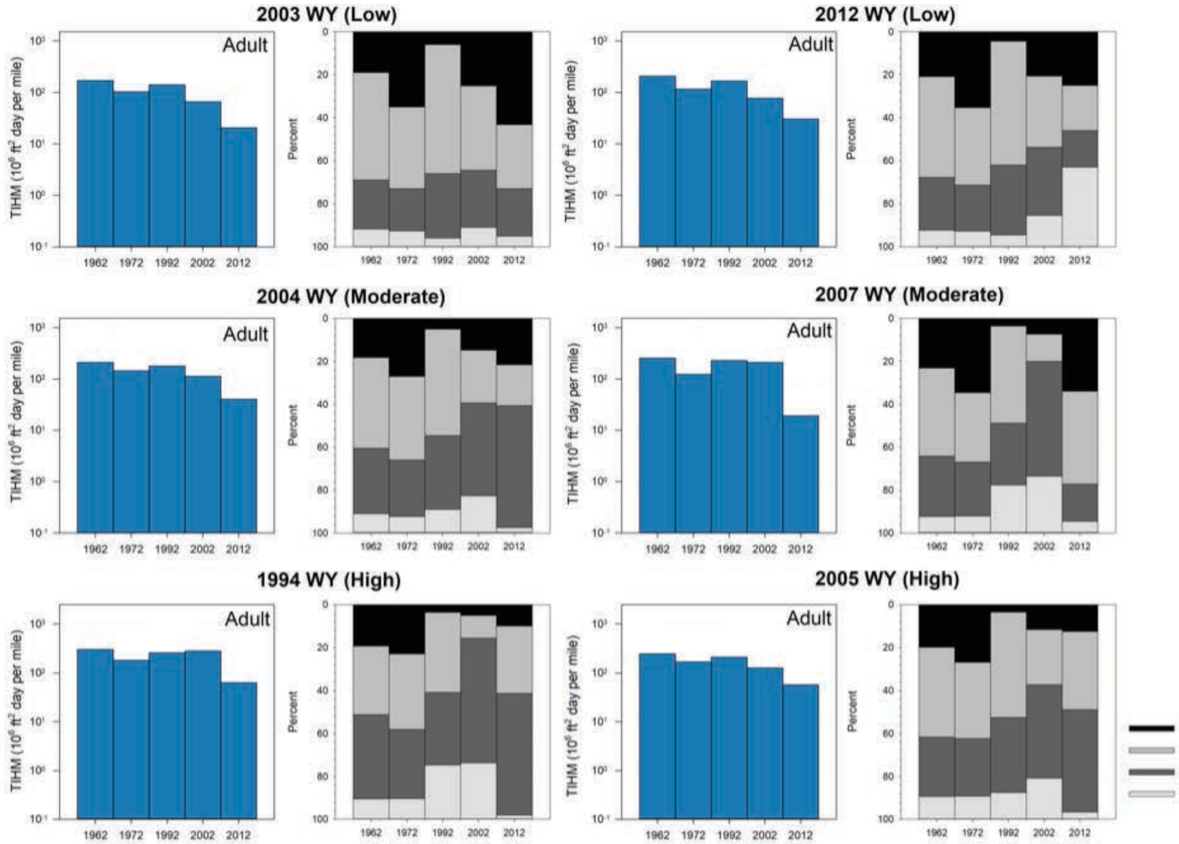



Figure 33. Time Integrated Habitat Metrics (TIHMs; colored bars) for the adult life-stage and percentage contribution by subreach (stacked bars) calculated using selected annual hydrographs and available channel geometries to assess the effects of geomorphic changes over time.

### *Channel-Habitat Evolution Models*

Stages of channel evolution were identified in the San Acacia Reach based on designations for the Middle Rio Grande (Table 6, Figure 34; 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 and spatial variation in planform evolution stages in the study area. The San Acacia Reach showed variable trends in channel evolutions that were best described at the subreach scale. Although most subreaches were characterized by the initial stages of the model in 1962 (Stage 1–3), subreaches evolved differently and display distinct spatial trends during the study period.

The San Acacia subreach (SA1–SA4) was characterized by a gradual progression to stages M4–M5, which indicate excessive transport capacity. The evolution of this subreach is described by narrowing and incising of the channel 1992–2012, thereby increasing bankfull discharge and reduced habitat availability across discharges (Figure 35). The Escondida subreach (E1–E5) was characterized by a gradual transition from excessive sediment transport capacity (E1) to “equilibrium” sediment transport capacity (E2–E3) to deficient sediment transport capacity (E4–E5) over the length of the subreach (Figure 36). This transition was captured by the stages of the model in 2012, which showed a downstream trend from stage M4 (E1) to stage 3 (E2–E3) to A4 (E4–E5). The Bosque del Apache subreach was characterized by a gradual progression through the initial stages of the model 1962–2002 with a subreach-wide transition to the aggrading stages of the model throughout the subreach by 2012 (Figure 37). Evidence of aggradation can be seen in cross-sections of this subreach prior to 2012, however, the model of planform evolution did not capture this trend until 2012. Of note, a sediment plug formed in this reach in 2008 (subreach BDA3) due to rapid aggradation – although this event is described by stage A5 of the model, stage classifications were only made for survey years and therefore this designation was not included. The Elephant Butte subreach (EB1–EB5) exhibited the most temporally variable morphological trends due to its proximity to Elephant Butte Reservoir (Figure 38) – changes in pool elevation at the reservoir affect sediment transport conditions upstream (Holste, 2015). Overall, this subreach experienced a shift to deficient transport capacity, a period during which reservoir levels increased (1972–1992) and remained elevated (1992–2002), followed by a shift to excessive transport capacity 2002–2012, a period during which reservoir levels rapidly receded and remained relatively low (Figure 39). The morphological trends observed in the Elephant Butte subreach show the importance of considering the influence that external factors can exert on channel evolution processes.

Stages of the planform evolution model were further evaluated using representative cross-sections, aerial imagery, and flow-habitat curves (Figures 35–38; Figures A-2–A-20). 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. 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 flow-habitat curves among subreaches revealed spatially distinct trends in channel evolution in the San Acacia Reach. These trends included: 1) a transition from excessive sediment transport capacity to deficient sediment transport capacity in the Escondida subreach, 2) increased bankfull discharge, loss of floodplain connectivity, and decreased habitat availability over time in the San Acacia subreach, 3) maintenance of floodplain connectivity and habitat availability in the Escondida and Bosque del Apache subreaches associated with relatively stable bankfull discharges over time, and 4) temporally variable channel evolution patterns in the Elephant Butte subreach related to changes in reservoir level. Spatially distinct trends are illustrated for representative subreaches in this section (Figures 35–38; refer to Appendix A for channel-habitat evolution models by subreach. These patterns observed at the subreach scale indicate spatially variable responses to channel evolution within the San Acacia Reach. At the reach-scale (i.e., San Acacia Reach), floodplain connectivity and habitat availability have remained relatively high, however, this reach-scale trend is not spatially uniform (i.e., subreaches show varying degrees of change to floodplain connectivity and habitat availability over the study period).

Table 6. Classification of San Acacia subreaches based on the planform evolution model for the Middle Rio Grande (Massong et al., 2010).

| Subreach | 1962 | 1972 | 1992 | 2002 | 2012 |
|---|---|---|---|---|---|
| SA1 | 1 | 1 | 1 | 3 | M4 |
| SA2 | 1 | 1 | 3 | M4 | M5 |
| SA3 | 1 | 1 | 1 | 2 | 3 |
| SA4 | 1 | 1 | M4 | M5 | M5 |
| E1 | 1 | 1 | 2 | M4 | M5 |
| E2 | 1 | 1 | 2 | 2 | 3 |
| E3 | 1 | 1 | 2 | 3 | 3 |
| E4 | 1 | 1 | 2 | 2 | A4 |
| E5 | 1 | 1 | 2 | 3 | A4 |
| BDA1 | 1 | 1 | 3 | A4 | A4 |
| BDA2 | 1 | 1 | 2 | 2 | A4 |
| BDA3 | 1 | 1 | 2 | 3 | A4^ |
| BDA4 | 1 | 1 | 3 | 3 | A4 |
| BDA5 | 3 | 3 | A4 | A4 | A4 |
| EB1 | 1 | 1 | A4 | A4 | M4 |
| EB2 | 1 | 1 | A4 | A4 | M4 |
| EB3 | A4 | A4 | A6 | A4^ | M4 |
| EB4 | M4 | M5 | 3 | A4 | M4 |
| EB5 | 3 | 3 | A4 | A4 | M4 |

^ Sediment plugs formed in BDA3 in 2008 and 2019, and EB3 in 2005.

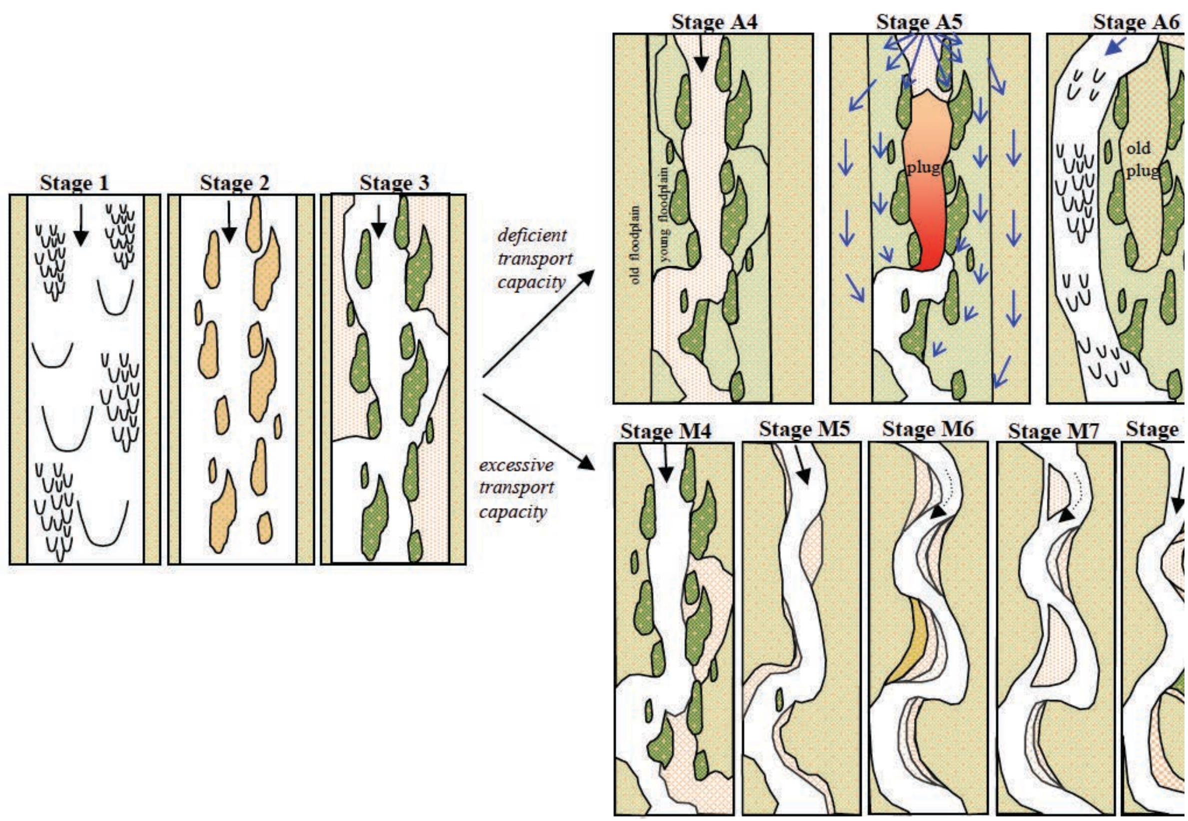
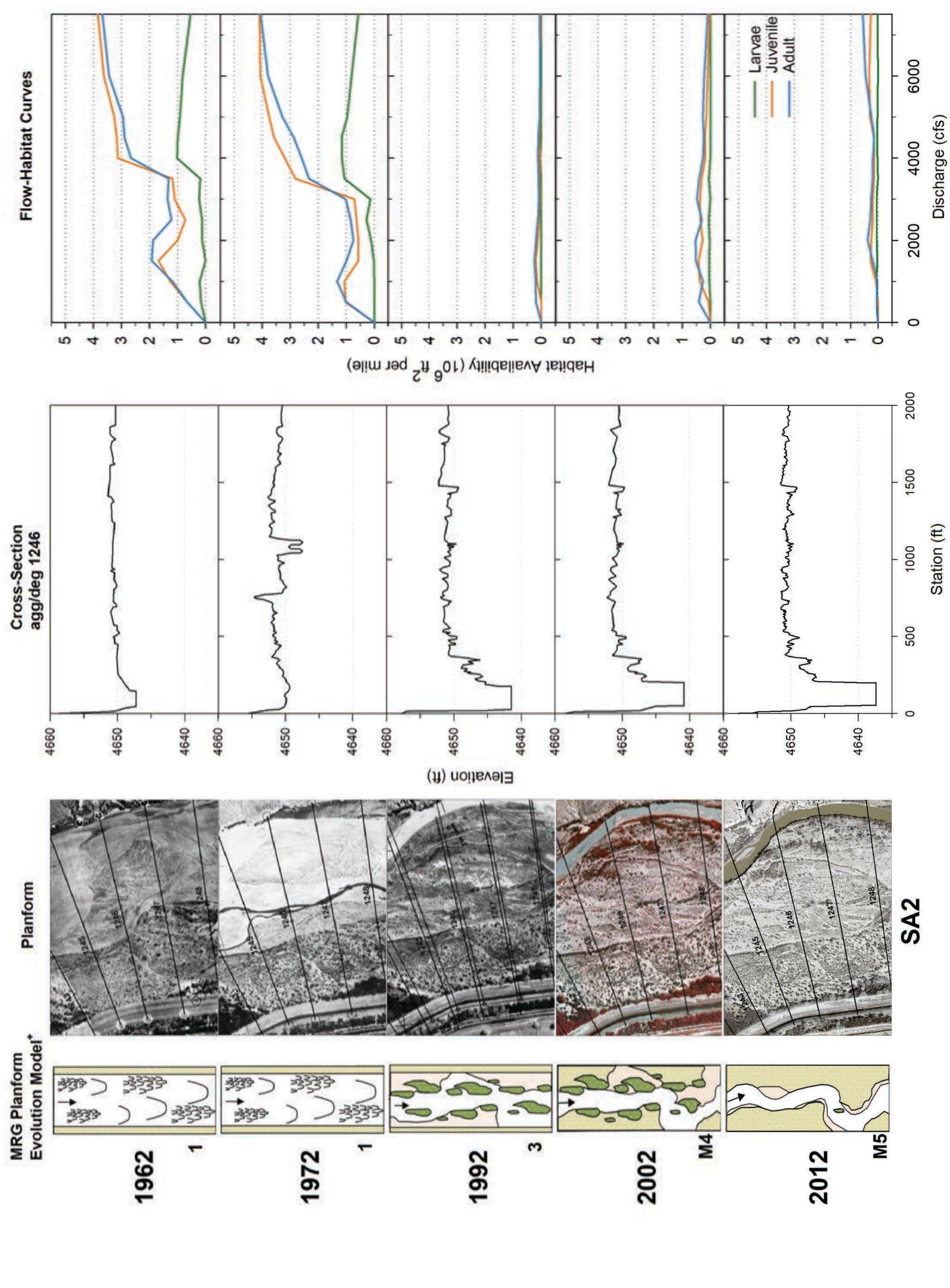
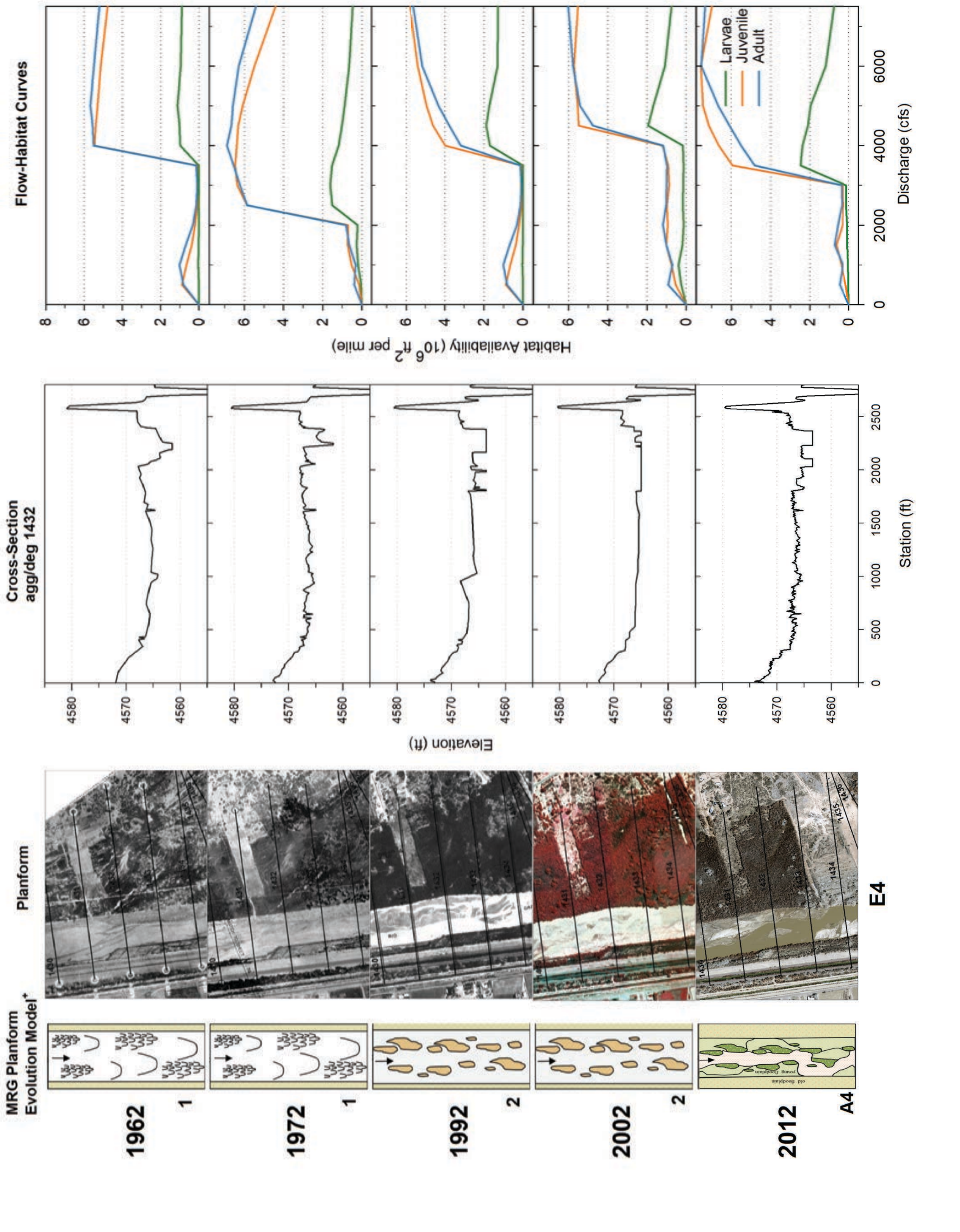
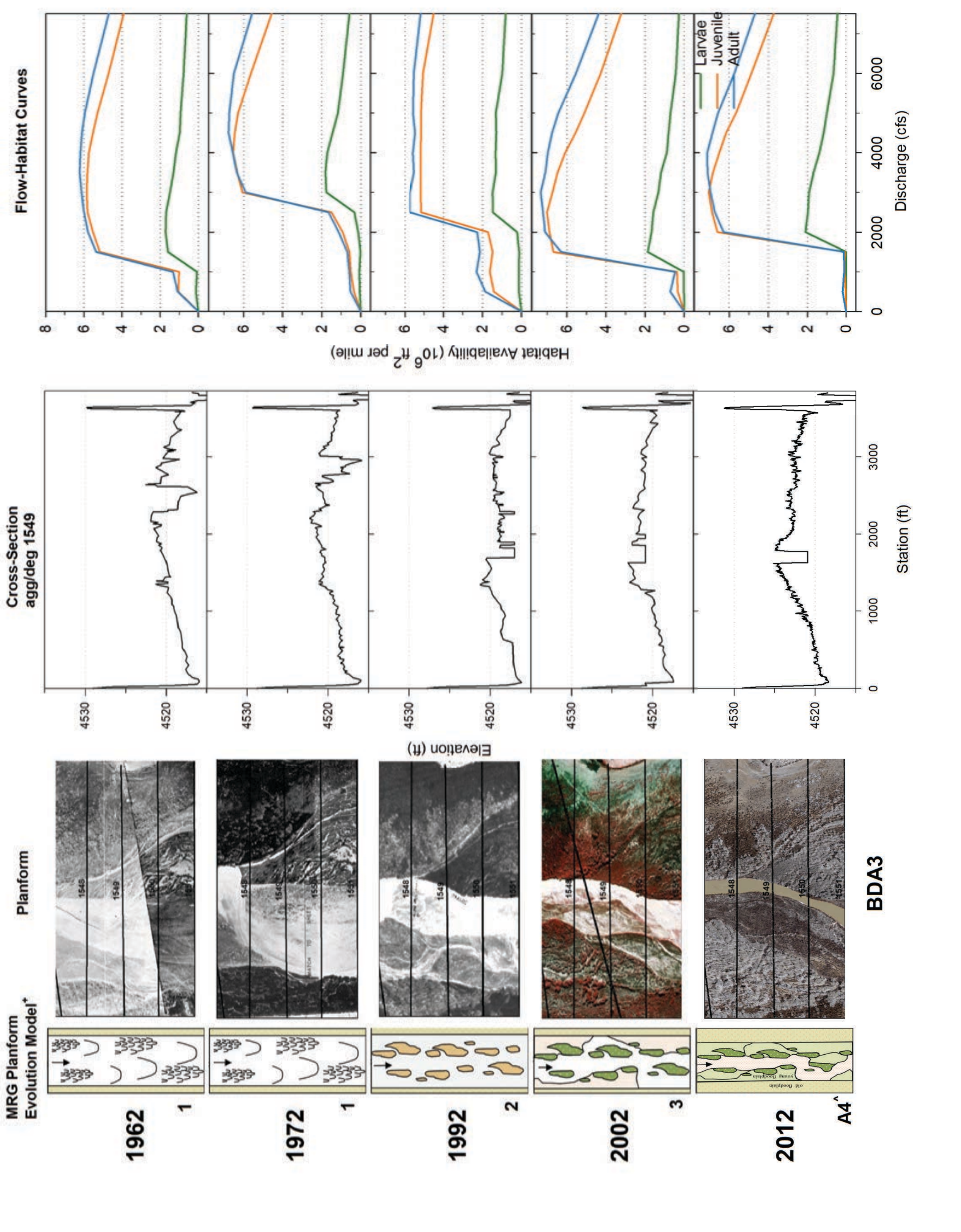
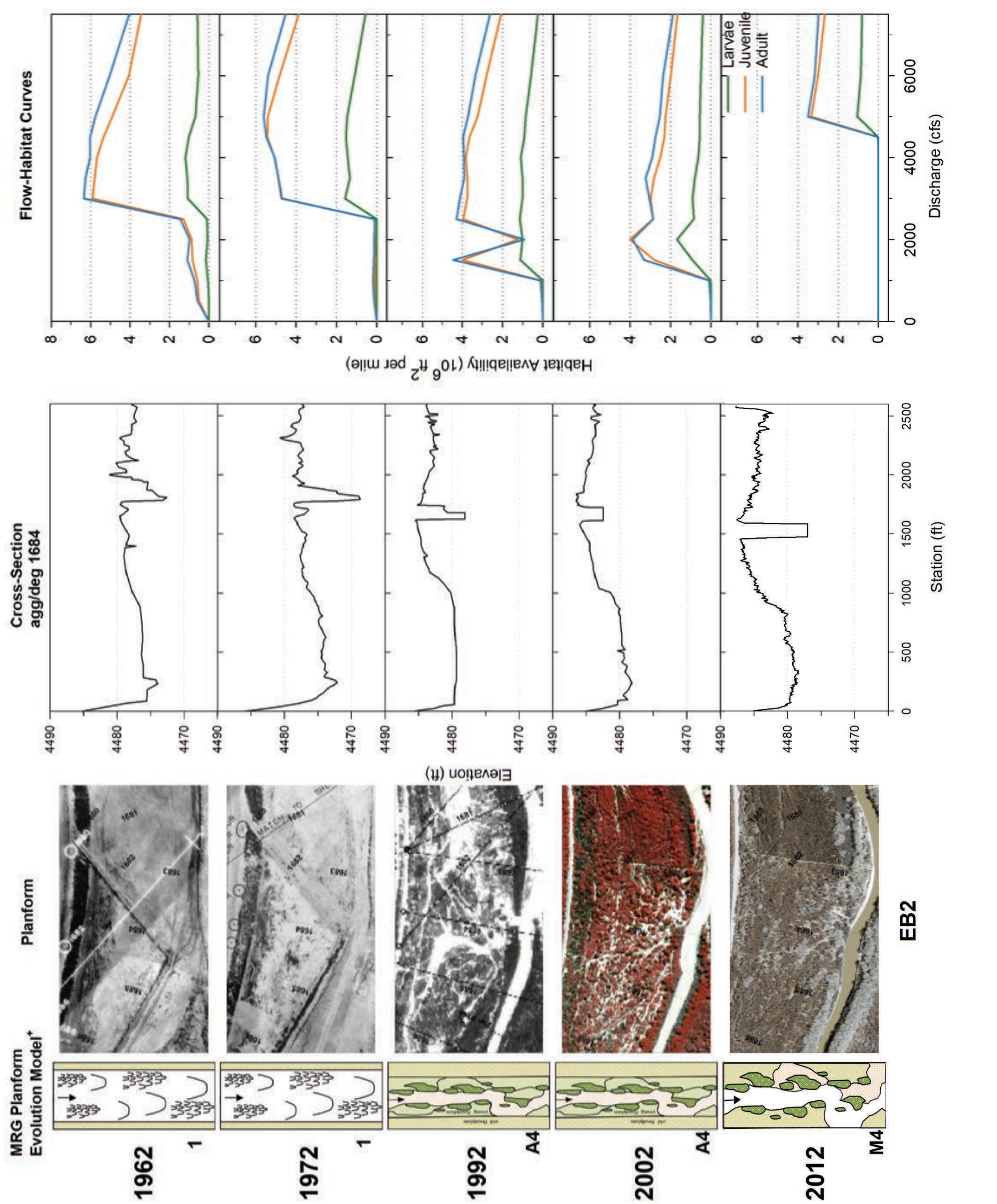



Figure 34. Planform evolution model for the Middle Rio Grande (from Massong et al., 2010).


MRG Planform Evolution Model+
Planform
Cross-Section agg/deg 1246
Flow-Habitat Curves
1962
1
1972
1
1992
3
2002
M4
2012
M5
SA2
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


MRG Planform Evolution Model+
Planform
Cross-Section agg/deg 1432
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
2
2012
A4
E4
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1549
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
3
2012
A4
BDA3
Elevation (ft)
Station (ft)
Habitat Availability ($10^6$ ft$^2$ per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1684
Flow-Habitat Curves
1962
1972
1992
2002
2012
1
1
A4
A4
M4
EB2
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

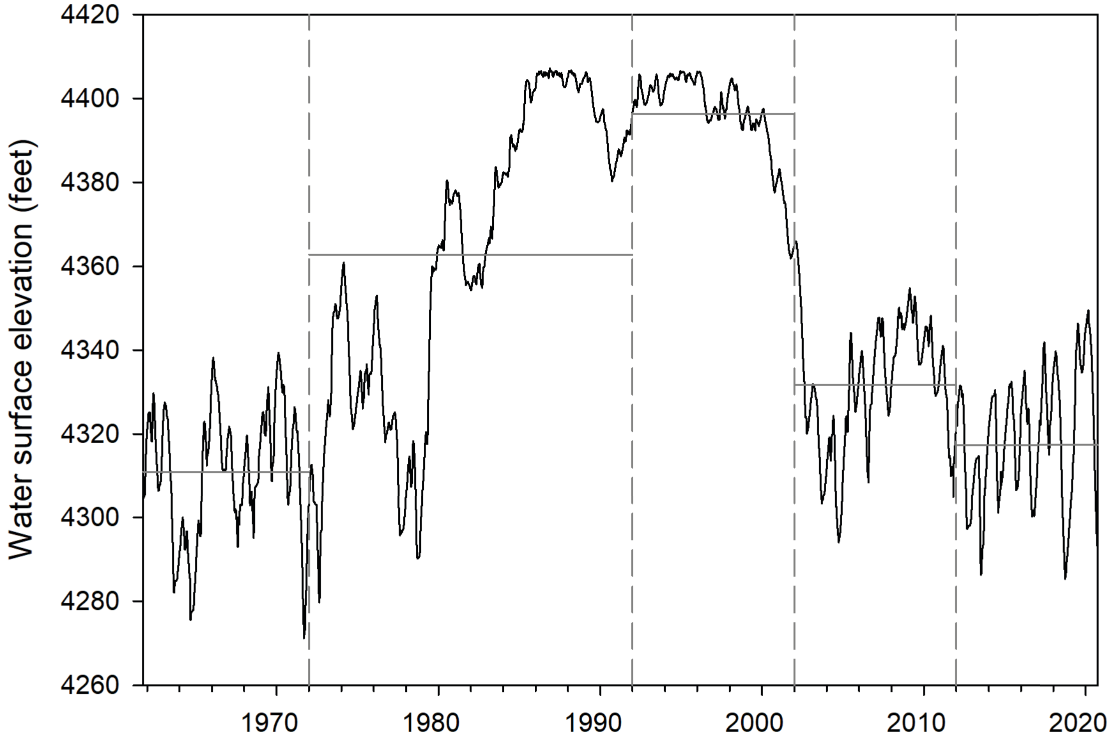



Figure 39. Water surface elevation at Elephant Butte Reservoir 1962–2021. Solid black line represents water surface elevation, dashed gray lines (vertical) represent channel survey datasets, and solid gray lines (horizontal) represent mean water surface elevation between channel surveys.

*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 San Acacia Reach. Rio Grande Silvery Minnow densities ($E(x)$; estimated using October sampling-site data [1993–2021]) were generated from the year model ($\delta$[Year] $\mu$[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 40). However, recent monitoring efforts revealed a substantial decrease (–96.9%) in the density of Rio Grande Silvery Minnow from 2019 ($E(x) = 2.23$) to 2020 ($E(x) = 0.07$), and its density remained relatively low in 2021 ($E(x) = 0.52$). 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 41). Estimates of $E(x)$ were generally highest when habitat and flow values were elevated (Figure 42: A–I). For brevity, trends in reach-scale estimated density are discussed herein; reach-scale occurrence probability ($\delta$) and lognormal density ($\mu$) for the study period are presented in the following section.

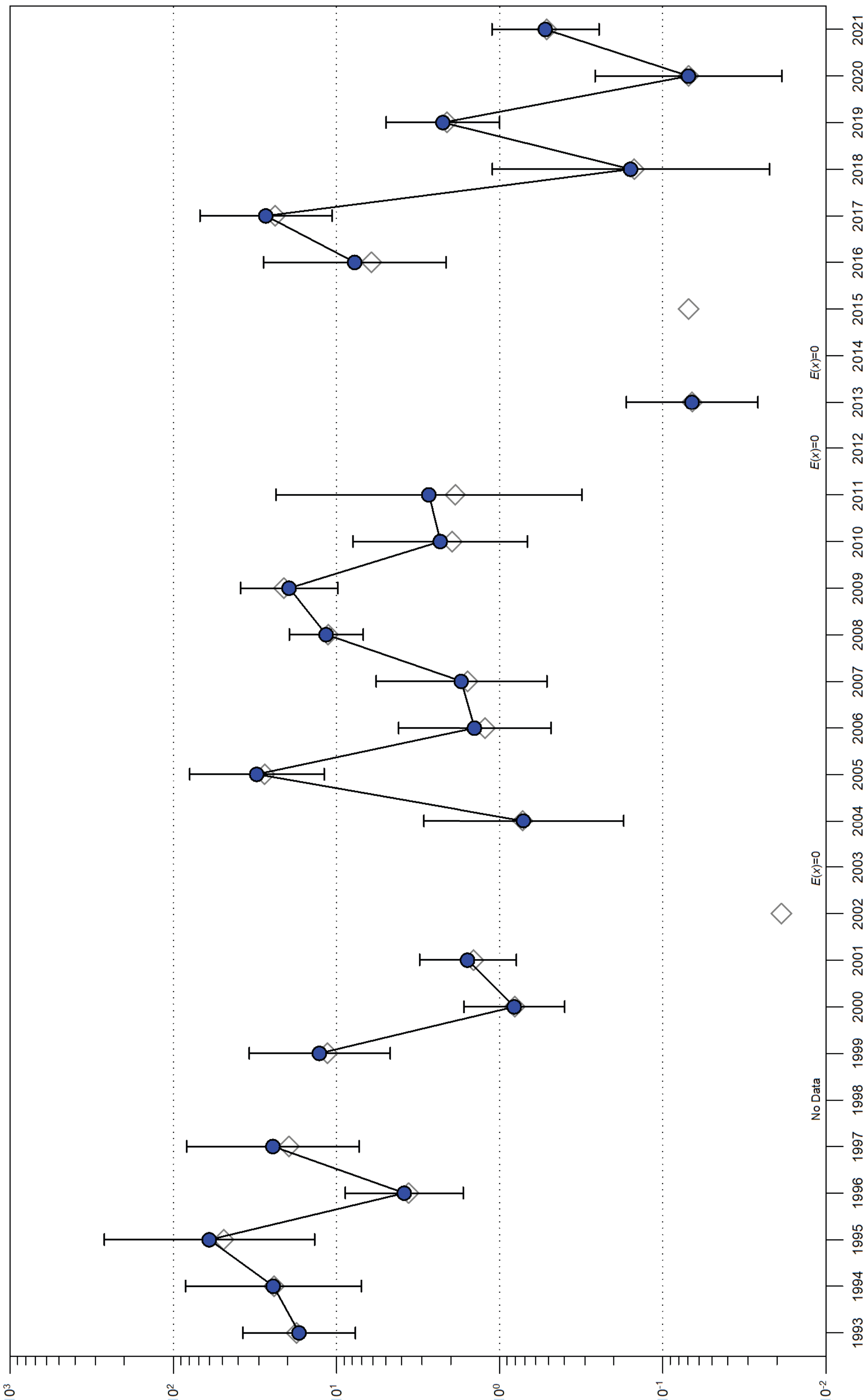
*Long-Term Ecological Relationships*

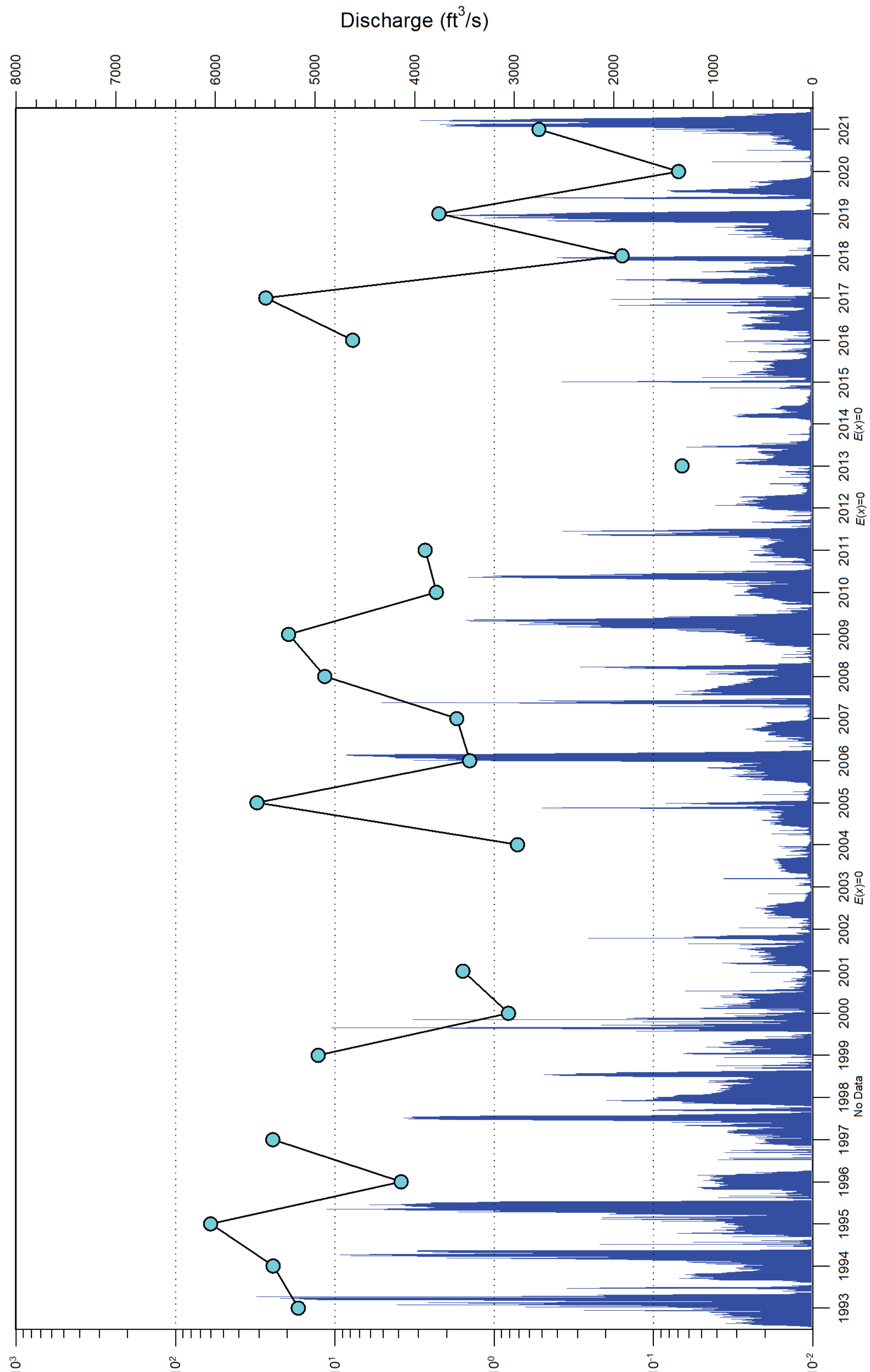
Relationships between Rio Grande Silvery Minnow population parameters and environmental covariates (i.e., habitat and flow metrics; Tables 5 and 7) were evaluated using robust statistical methods. Habitat and flow metrics were evaluated independently (Table 8, 9) and together (Table 10). The occurrence probability ($\delta$) and the lognormal density ($\mu$), estimated from the year model ($\delta$[Year] $\mu$[Year]), were closely associated with environmental covariates over time (1993–2021). Estimates of $\delta$ increased with elevated habitat and flow values (Figures 43–44). Similar and consistent results were obtained for relationships between $\mu$ and habitat/flow metrics (Figures 45–46).

Generalized linear models of Rio Grande Silvery Minnow mixture-model estimates revealed that variation in both $\delta$ and $\mu$ was strongly predicted by changes in habitat metrics across years (1993–2021 [TIHMs only]; Table 8). The top ecological model ($\delta$[MayJunHab+$R$] $\mu$[MayJunHab+$R$]) received 77.6% of the AIC$_c$ weight ($w_i$) out of the 64 models considered. The top $\delta$ covariate (MayJunHab) accounted for 45.9% of the deviance ($P < 0.001$) explained by the $\delta$(Year) model over the $\delta$(.) model. We also found significant effects for JulSepHab (35.5%; $P < 0.001$) and OctAprHab (31.3%; $P < 0.01$). Further, the top $\mu$ covariate (MayJunHab) accounted for 41.1% of the deviance ($P < 0.001$) explained by the $\mu$(Year) model over the $\mu$(.) model. We also found a significant effect for OctAprHab (18.6%; $P < 0.05$), but not for JulSepHab (2.7%). The top two habitat models, which accounted for nearly all the cumulative $w_i$ (ca. 97%), were based on metrics representing elevated larval (May–June) and juvenile (July–September) habitat availability. In summary, more larval fish habitat during spring best predicted the increased occurrence and density of Rio Grande Silvery Minnow in October over time.

Similarly, generalized linear models of Rio Grande Silvery Minnow mixture-model estimates revealed that variation in both $\delta$ and $\mu$ was also strongly predicted by changes in flow metrics across years (1993–2021 [flow metrics only]; Table 9). The top ecological model ($\delta$[MayJun28dHigh+$R$] $\mu$[MayJun28dHigh+$R$]) received 51.6% of the AIC$_c$ weight ($w_i$) out of the 196 models considered. The top $\delta$ covariate (MayJun28dHigh) accounted for 47.1% of the deviance ($P < 0.001$) explained by the $\delta$(Year) model over the $\delta$(.) model. We also found significant effects for MayJunMean (46.6%; $P < 0.001$), OctAprMean (21.6%; $P < 0.05$), and JulSepMean (18.7%; $P < 0.05$), but not for OctApr7dLow (3.2%) or JulSep7dLow (2.2%). Further, the top $\mu$ covariate (MayJun28dHigh) accounted for 50.0% of the deviance ($P < 0.001$) explained by the $\mu$(Year) model over the $\mu$(.) model. We also found significant effects for MayJunMean (45.5%; $P < 0.001$), OctAprMean (36.7%; $P < 0.001$), and OctApr7dLow (14.5%; $P < 0.05$), but not for JulSepMean (2.5%) or JulSep7dLow (1.4%). The top six flow models, which accounted for nearly all the cumulative $w_i$ (> 99%), 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 in October over time.

Finally, generalized linear models of Rio Grande Silvery Minnow mixture-model estimates revealed that variation in both $\delta$ and $\mu$ was better predicted by changes in flow metrics, as compared to habitat metrics, across years (1993–2021 [flow and habitat metrics]; Table 10). The top ecological model ($\delta$[MayJun28dHigh+*R*] $\mu$[MayJun28dHigh+*R*]) received 40.2% of the $AIC_c$ weight ($w_i$) out of the 400 models considered. The top two models, which accounted for most of the cumulative $w_i$ (ca. 66%), 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 in October over time.


1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
No Data
E(x)=0
E(x)=0
E(x)=0
$10^3$
$10^2$
$10^1$
$10^0$
$10^{-1}$
$10^{-2}$


Discharge (ft$^3$/s)
8000
7000
6000
5000
4000
3000
2000
1000
0
1993
1994
1995
1996
1997
1998
No Data
1999
2000
2001
2002
2003
E(x)=0
2004
2005
2006
2007
2008
2009
2010
2011
2012
E(x)=0
2013
2014
E(x)=0
2015
2016
2017
2018
2019
2020
2021
10$^3$
10$^2$
10$^1$
10$^0$
10$^{-1}$
10$^{-2}$

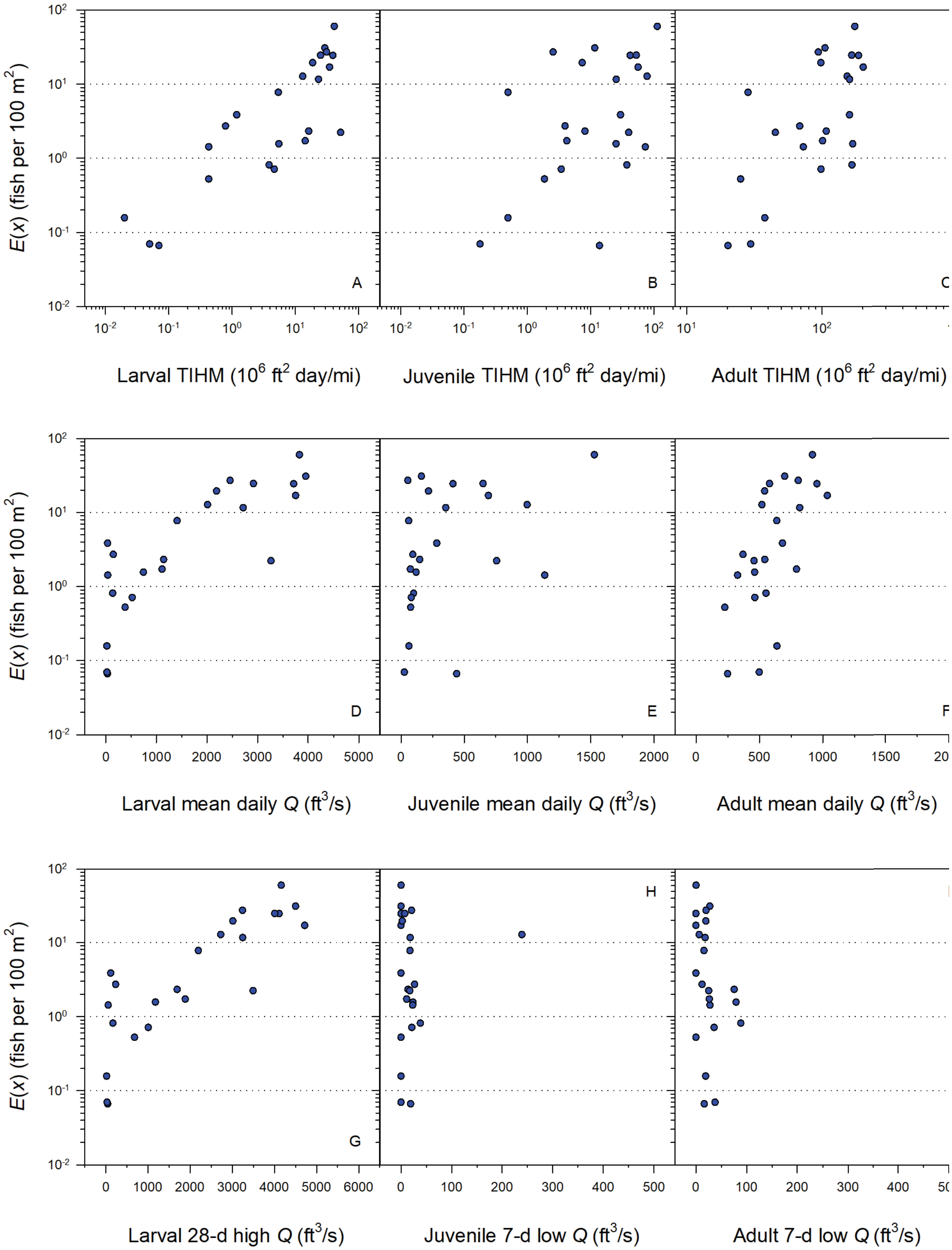
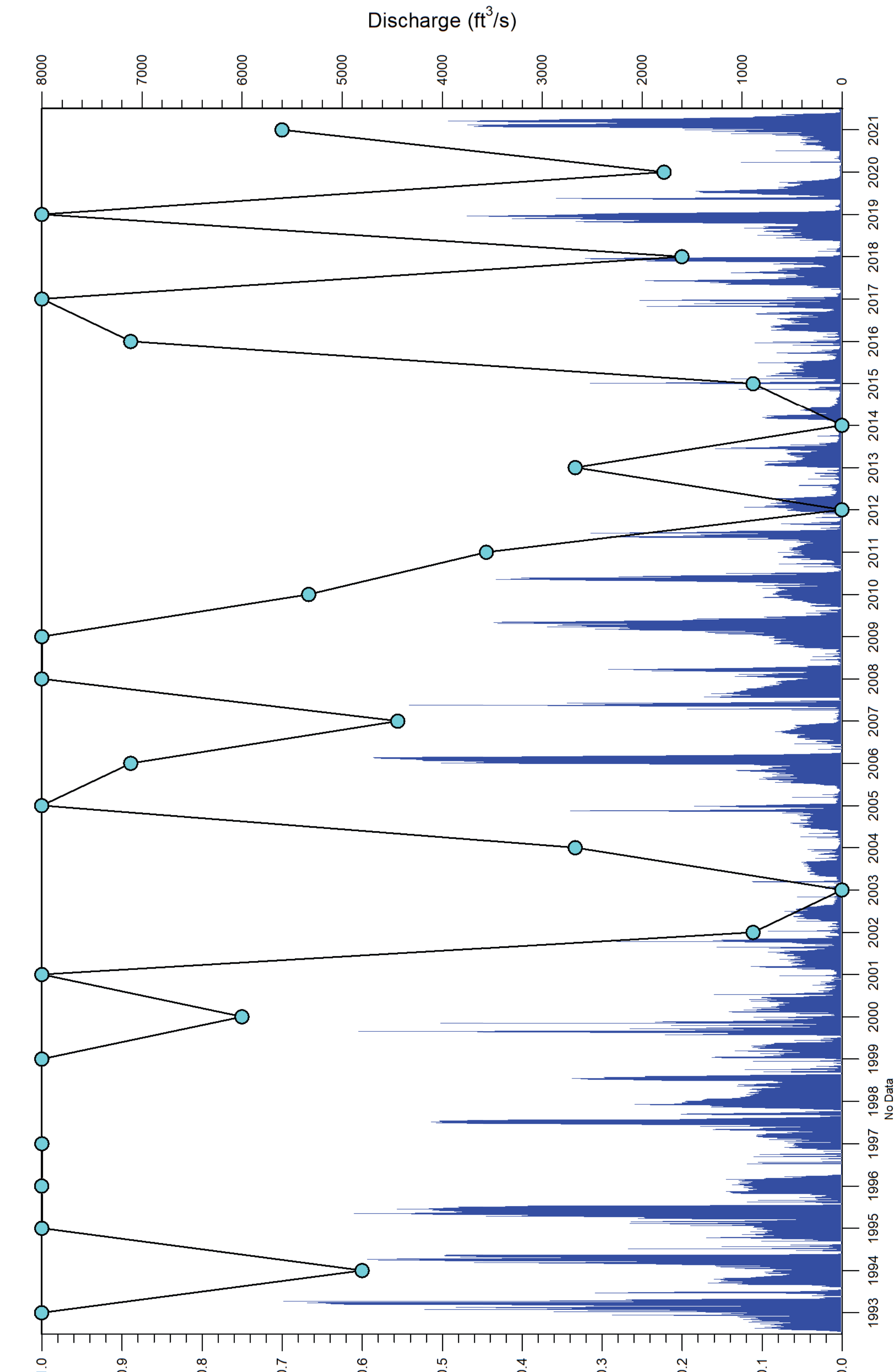



Figure 42. Bivariate plots of densities of the Rio Grande Silvery Minnow ($E(x)$; estimated using October sampling-site data), habitat data (panels A–C), and flow data (panels D–I).

Table 7. Flow metrics for the San Acacia Reach 1993–2021. Discharge data for the San Acacia Reach were obtained from the Rio Grande floodway at San Marcial, NM (USGS 08358400).

| | San Acacia 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 | 3753.2 | 691.2 | 1037.2 | 4720.0 | 0.0 | 0.0 |
| 1994 | 3714.8 | 411.6 | 954.3 | 4114.6 | 0.0 | 0.0 |
| 1995 | 3828.7 | 1530.2 | 920.0 | 4162.9 | 0.0 | 0.0 |
| 1996 | 38.4 | 283.6 | 682.3 | 117.7 | 0.0 | 0.0 |
| 1997 | 2919.9 | 649.0 | 580.2 | 4007.9 | 7.4 | 0.1 |
| 1998 | 1632.7 | 212.7 | 1074.5 | 2412.5 | 0.0 | 577.4 |
| 1999 | 2013.3 | 997.7 | 519.8 | 2727.9 | 239.1 | 6.5 |
| 2000 | 136.7 | 100.2 | 552.1 | 173.6 | 38.4 | 88.7 |
| 2001 | 745.2 | 120.0 | 462.5 | 1179.3 | 23.6 | 79.3 |
| 2002 | 54.4 | 141.2 | 272.8 | 94.0 | 13.0 | 36.9 |
| 2003 | 110.6 | 64.7 | 228.9 | 157.1 | 17.4 | 29.7 |
| 2004 | 524.3 | 81.4 | 464.7 | 1010.8 | 21.5 | 36.0 |
| 2005 | 3951.6 | 161.9 | 700.4 | 4503.9 | 0.0 | 27.3 |
| 2006 | 41.0 | 1136.5 | 327.9 | 59.4 | 23.1 | 27.7 |
| 2007 | 1113.2 | 73.9 | 794.1 | 1888.2 | 11.1 | 26.4 |
| 2008 | 2717.4 | 354.0 | 818.6 | 3248.2 | 18.1 | 18.2 |
| 2009 | 2191.0 | 218.0 | 541.9 | 3015.0 | 2.5 | 19.5 |
| 2010 | 1145.8 | 148.3 | 542.6 | 1694.0 | 13.7 | 75.6 |
| 2011 | 148.8 | 93.0 | 370.7 | 237.0 | 26.7 | 12.1 |
| 2012 | 110.1 | 52.0 | 475.4 | 185.1 | 18.9 | 28.8 |
| 2013 | 35.6 | 440.6 | 249.8 | 49.1 | 19.1 | 16.6 |
| 2014 | 210.8 | 231.0 | 393.7 | 367.0 | 22.1 | 56.0 |
| 2015 | 831.0 | 350.1 | 440.7 | 1236.6 | 14.6 | 28.9 |
| 2016 | 1408.9 | 60.0 | 637.6 | 2196.1 | 17.7 | 16.0 |
| 2017 | 2458.1 | 53.9 | 807.0 | 3246.1 | 20.8 | 20.0 |
| 2018 | 21.9 | 61.7 | 638.4 | 23.5 | 0.0 | 19.4 |
| 2019 | 3264.6 | 757.5 | 457.7 | 3496.4 | 17.4 | 25.2 |
| 2020 | 26.0 | 26.2 | 499.3 | 32.8 | 0.0 | 37.7 |
| 2021 | 384.3 | 75.5 | 226.7 | 683.2 | 0.0 | 0.0 |

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


Discharge (ft3/s)
No Data

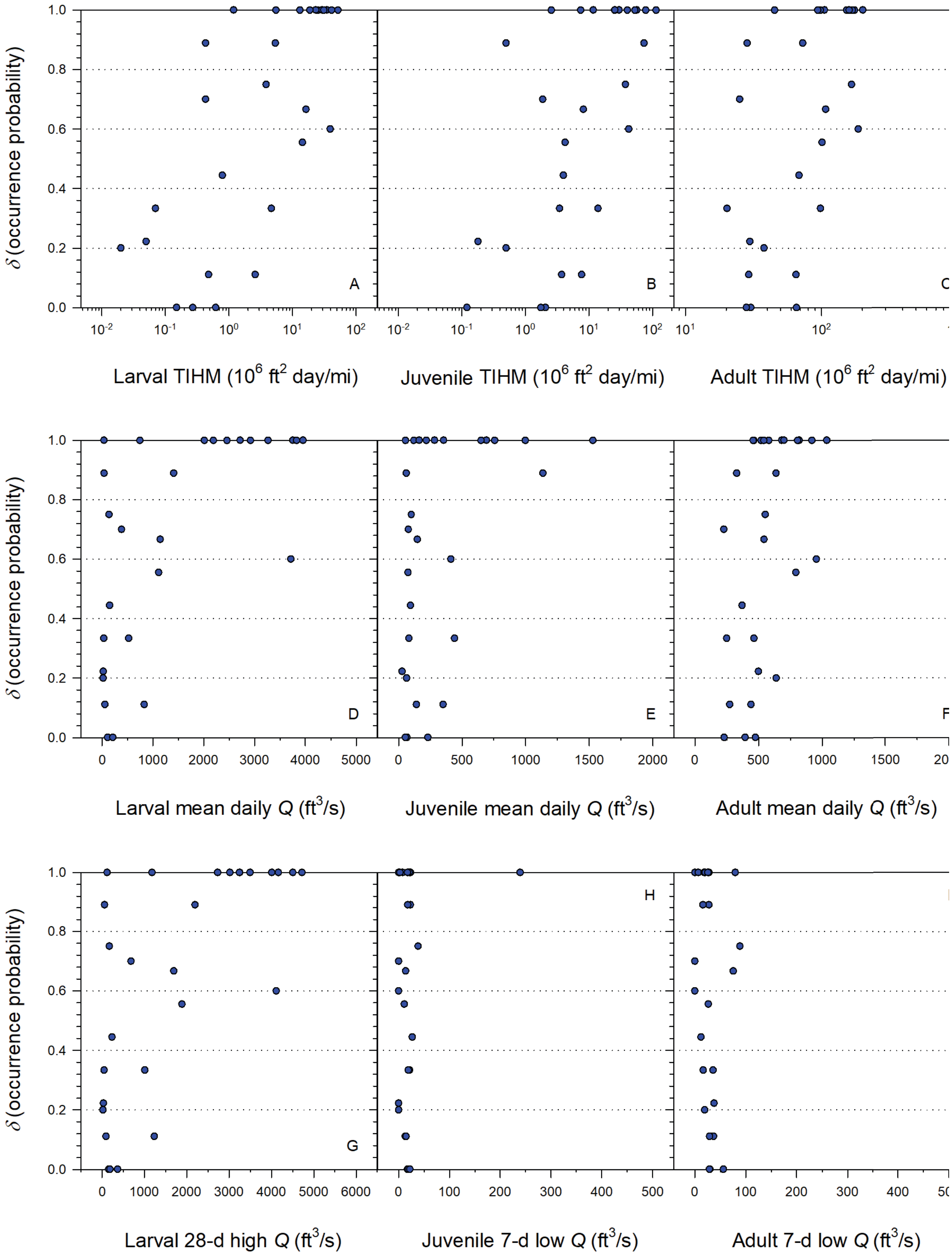
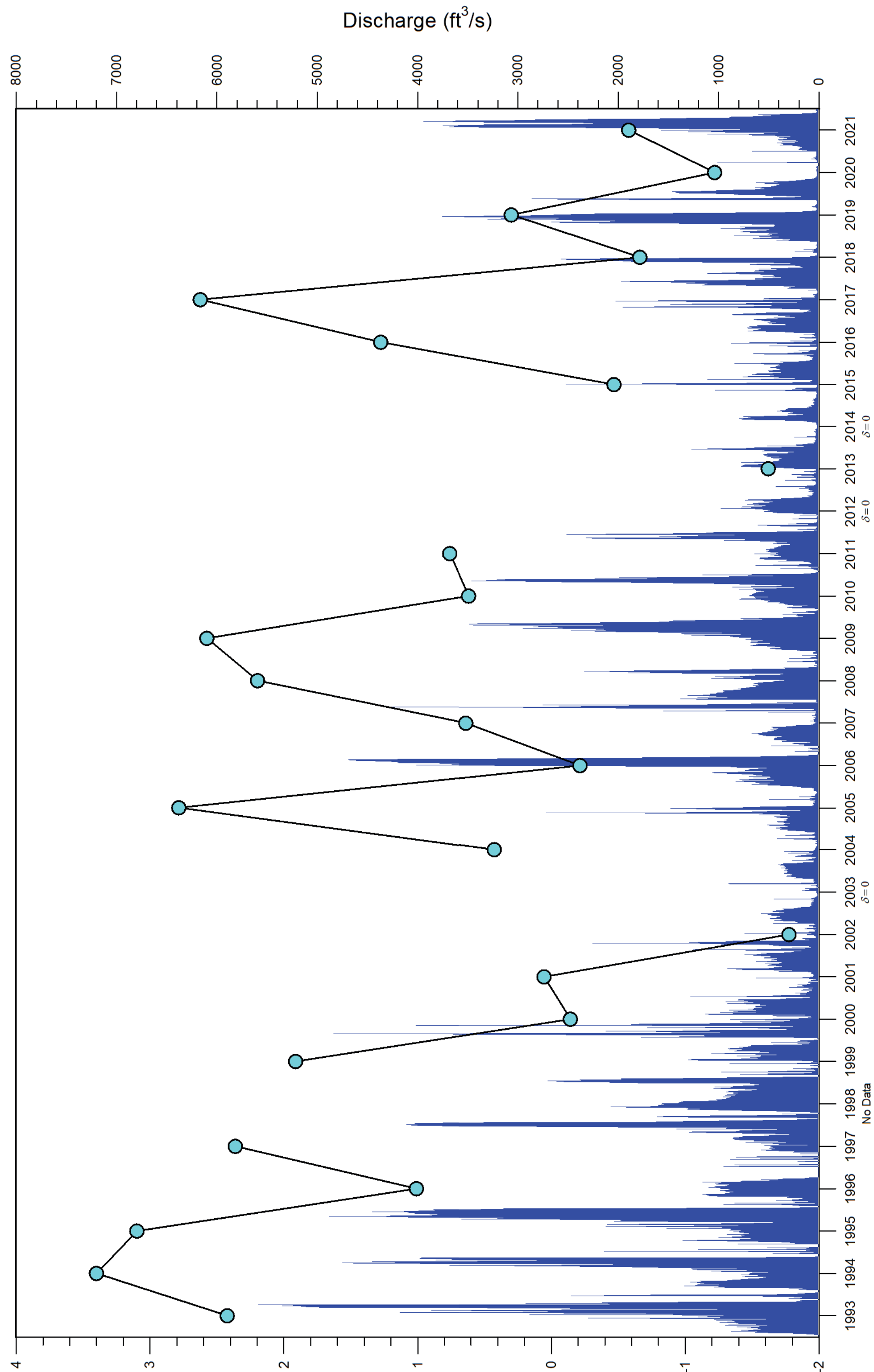



Figure 44. Bivariate plots of occurrence probabilities of the Rio Grande Silvery Minnow ($\delta$; estimated using October sampling-site data), habitat data (panels A–C), and flow data (panels D–I).


Discharge (ft3/s)
8000
7000
6000
5000
4000
3000
2000
1000
0
2021
2020
2019
2018
2017
2016
2015
2014
δ = 0
2013
2012
δ = 0
2011
2010
2009
2008
2007
2006
2005
2004
2003
δ = 0
2002
2001
2000
1999
1998
No Data
1997
1996
1995
1994
1993
4
3
2
1
0
-1
-2

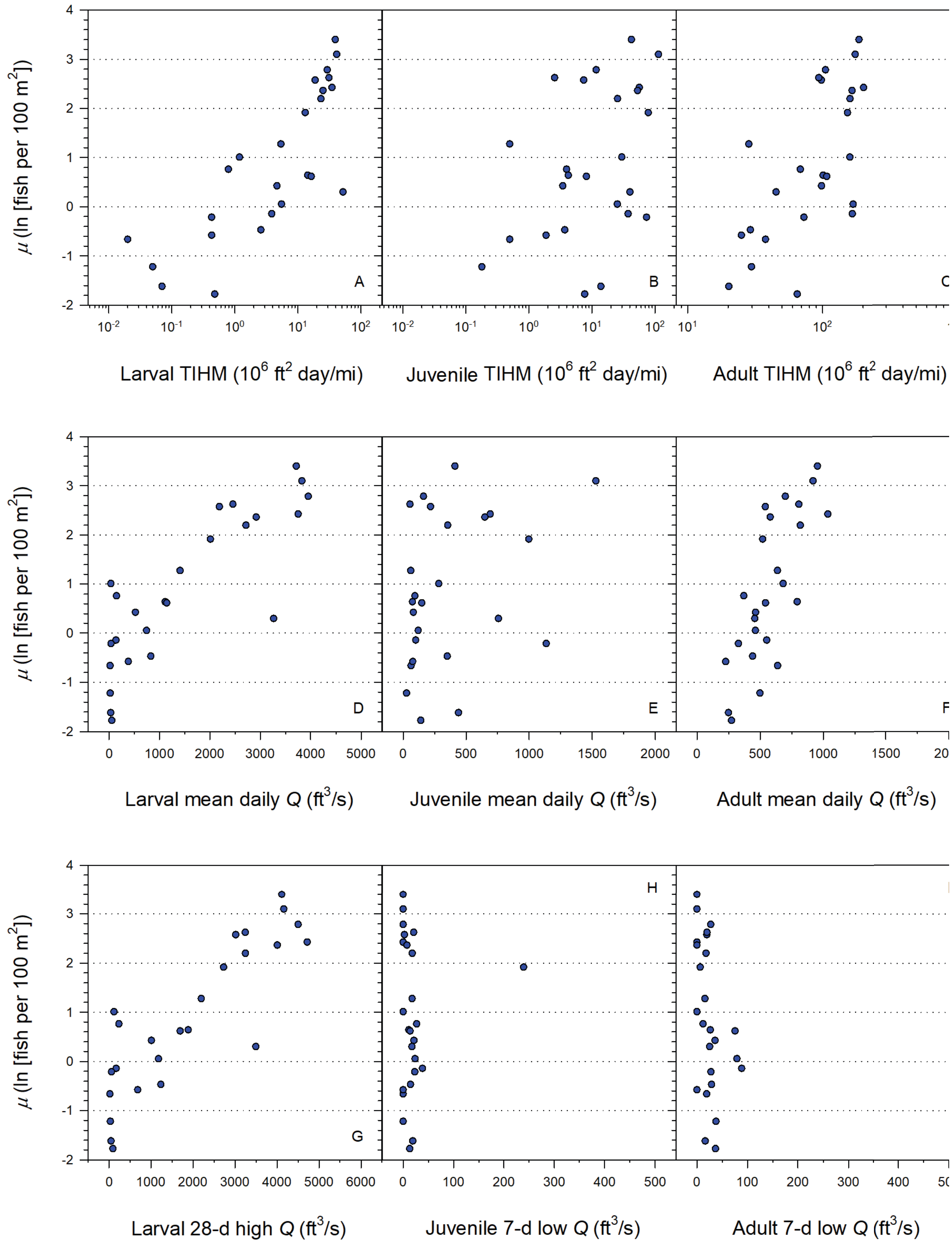



Figure 46. Bivariate plots of lognormal densities of the Rio Grande Silvery Minnow ($\mu$; estimated using October sampling-site data), habitat data (panels A–C), and flow data (panels D–I).

Table 8. Generalized linear models of mixture-model estimates for the Rio Grande Silvery Minnow, using October sampling-site data from the San Acacia Reach (1993–2021). Only habitat metrics (MayJunHab, JulSepHab, and OctAprHab) were included.

| **Model**[1] | **logLike**[2] | **K**[3] | **AIC$_c$**[4] | **$w_i$**[4] |
|---|---|---|---|---|
| $\delta$(MayJunHab+*R*) $\mu$(MayJunHab+*R*) | 402.21 | 9 | 420.99 | 0.7763 |
| $\delta$(JulSepHab+*R*) $\mu$(MayJunHab+*R*) | 404.99 | 9 | 423.78 | 0.1932 |
| $\delta$(OctAprHab+*R*) $\mu$(MayJunHab+*R*) | 409.79 | 9 | 428.57 | 0.0176 |
| $\delta$(JulSepHab+*R*) $\mu$(OctAprHab+*R*) | 410.73 | 9 | 429.51 | 0.0110 |
| $\delta$(OctAprHab+*R*) $\mu$(OctAprHab+*R*) | 415.68 | 9 | 434.46 | 0.0009 |
| $\delta$(MayJunHab+*R*) $\mu$(OctAprHab+*R*) | 417.69 | 9 | 436.47 | 0.0003 |
| $\delta$(*R*) $\mu$(MayJunHab+*R*) | 420.31 | 8 | 436.93 | 0.0003 |
| $\delta$(JulSepHab+*R*) $\mu$(JulSepHab+*R*) | 419.56 | 9 | 438.34 | 0.0001 |
| $\delta$(MayJunHab+*R*) $\mu$(MayJunHab) | 424.30 | 7 | 438.78 | 0.0001 |
| $\delta$(JulSepHab+*R*) $\mu$(*R*) | 426.76 | 7 | 441.24 | <0.0001 |

[1] = Models included all $\delta$ and $\mu$ combinations of null effects (.), random effects (*R*), and 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 by Akaike's information criterion (AIC$_c$) and include the AIC$_c$ weight ($w_i$).

Table 9. Generalized linear models of mixture-model estimates for the Rio Grande Silvery Minnow, using October sampling-site data from the San Acacia Reach (1993–2021). Only flow metrics (MayJunMean, JulSepMean, OctAprMean, MayJun28dHigh, JulSep7dLow, and OctApr7dLow) were included.

| **Model**[1] | **logLike**[2] | **K**[3] | **AIC$_c$**[4] | ***w$_i$***[4] |
|---|---|---|---|---|
| $\delta$(MayJun28dHigh+*R*) $\mu$(MayJun28dHigh+*R*) | 394.73 | 9 | 413.52 | 0.5159 |
| $\delta$(MayJunMean+*R*) $\mu$(MayJun28dHigh+*R*) | 359.66 | 9 | 414.44 | 0.3250 |
| $\delta$(MayJun28dHigh+*R*) $\mu$(MayJunMean+*R*) | 398.56 | 9 | 417.35 | 0.0760 |
| $\delta$(MayJunMean+*R*) $\mu$(MayJunMean+*R*) | 399.02 | 9 | 417.81 | 0.0604 |
| $\delta$(MayJun28dHigh+*R*) $\mu$(MayJun28dHigh) | 407.15 | 7 | 421.63 | 0.0089 |
| $\delta$(MayJunMean+*R*) $\mu$(MayJun28dHigh) | 408.34 | 7 | 422.82 | 0.0049 |
| $\delta$(JulSepMean+*R*) $\mu$(MayJun28dHigh+*R*) | 405.14 | 9 | 423.93 | 0.0028 |
| $\delta$(JulSepMean+*R*) $\mu$(MayJunMean+*R*) | 406.95 | 9 | 425.74 | 0.0011 |
| $\delta$(OctAprMean+*R*) $\mu$(MayJun28dHigh+*R*) | 406.97 | 9 | 425.75 | 0.0011 |
| $\delta$(MayJun28dHigh+*R*) $\mu$(OctAprMean+*R*) | 407.26 | 9 | 426.06 | 0.0010 |

[1] = Models included all $\delta$ and $\mu$ 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 by Akaike's information criterion (AIC$_c$) and include the AIC$_c$ weight ($w_i$).

Table 10. Generalized linear models of mixture-model estimates for the Rio Grande Silvery Minnow, using October sampling-site data from the San Acacia Reach (1993–2021). Habitat metrics (MayJunHab, JulSepHab, and OctAprHab) and flow metrics (MayJunMean, JulSepMean, OctAprMean, MayJun28dHigh, JulSep7dLow, and OctApr7dLow) were both included.

| **Model**[1] | **logLike**[2] | **K**[3] | **AIC$_c$**[4] | ***$w_i$***[4] |
|---|---|---|---|---|
| $\delta$(MayJun28dHigh+*R*) $\mu$(MayJun28dHigh+*R*) | 394.73 | 9 | 413.52 | 0.4023 |
| $\delta$(MayJunMean+*R*) $\mu$(MayJun28dHigh+*R*) | 395.66 | 9 | 414.44 | 0.2535 |
| $\delta$(MayJunHab+*R*) $\mu$(MayJun28dHigh+*R*) | 397.34 | 9 | 416.12 | 0.1096 |
| $\delta$(MayJun28dHigh+*R*) $\mu$(MayJunMean+*R*) | 398.56 | 9 | 417.35 | 0.0593 |
| $\delta$(MayJunMean+*R*) $\mu$(MayJunMean+*R*) | 399.02 | 9 | 417.81 | 0.0471 |
| $\delta$(JulSepHab+*R*) $\mu$(MayJun28dHigh+*R*) | 399.46 | 9 | 418.25 | 0.0378 |
| $\delta$(MayJunHab+*R*) $\mu$(MayJunMean+*R*) | 400.94 | 9 | 419.72 | 0.0181 |
| $\delta$(OctAprHab+*R*) $\mu$(MayJun28dHigh+*R*) | 401.38 | 9 | 420.16 | 0.0145 |
| $\delta$(JulSepHab+*R* h) $\mu$(MayJunMean+*R*) | 401.69 | 9 | 420.47 | 0.0125 |
| $\delta$(MayJunHab+*R*) $\mu$(MayJunHab+*R*) | 402.21 | 9 | 420.99 | 0.0096 |

[1] = Models included all $\delta$ and $\mu$ combinations of null effects (.), random effects (*R*), and habitat/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 by Akaike's information criterion (AIC$_c$) and include the AIC$_c$ weight ($w_i$).

# DISCUSSION

## *Comparison to Linkage Report I: Isleta Reach Analyses*

### *Hydrologic and Geomorphic Conditions*

Hydrologic conditions in the San Acacia Reach were generally characterized by lower flow magnitudes relative to the Isleta Reach (Figure 47d–f). Peak flow metrics (28d high May–June) were on average 30% lower in the San Acacia Reach; these flow metrics were lower in the San Acacia Reach for all years during the study period (1993–2019). Similarly, mean flow metrics corresponding to the principal life-stages of the Rio Grande Silvery Minnow were lower in the San Acacia Reach in 68 out of 81 the selected flow metrics 1993–2019. Low flow metrics (7d low July-September, October–April) showed an increase in the frequency of intermittent flow conditions in the San Acacia Reach – a total of 11 low flow metrics with zero values (i.e., 0 cfs) were recorded for the San Acacia Reach compared to a total of 2 in the Isleta Reach 1993–2019. 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. However, consistency in the selection of flow metrics in Linkage Reports I and II provides a basis for comparison of hydrologic conditions between the Isleta and San Acacia Reaches of the Middle Rio Grande.

Analysis of channel morphology over time in the San Acacia Reach indicated geomorphic trends not observed in the Isleta Reach. Notably, aggrading subreaches were present in the San Acacia Reach downstream of Escondida but were absent from the Isleta Reach. The occurrence of these channel stages provided evidence of the full range of geomorphic processes described by the Rio Grande planform evolution model that were not previously documented by this study (Massong et al., 2010; Mortensen et al., 2020). The upstream-most subreach, San Acacia, showed similar trends to subreaches of the Isleta Reach (e.g., I4, I5, P4), which were characterized by channel incision, increased bankfull discharge, and reduced floodplain connectivity. While geomorphic trends were relatively similar among subreaches in the Isleta Reach (e.g., increased bankfull discharge, reduced habitat availability at low to moderate flows), geomorphic trends in the San Acacia Reach were more spatially variable among subreach units. Additionally, the proximity of the downstream-most subreach to Elephant Butte Reservoir showed the importance of external factors on channel evolution processes. Such effects are not typically considered in conventional channel evolution models and explain temporally variable trends observed in the Elephant Butte subreach.

### *Habitat Conditions*

Flow-habitat curves generated for the San Acacia Reach differed from those from the Isleta Reach. For the initial survey years (1962 and 1972), flow-habitat curves showed similar bankfull discharges (1500–2500 cfs) and magnitudes of habitat availability (4–5 x $10^6$ $ft^2$ day/mi) in both the Isleta and San Acacia Reaches. For recent survey years (1992, 2002, and 2012), flow-habitat curves differed between Reaches. Whereas the Isleta Reach was 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. Habitat availability at the reach-scale was primarily contributed by the Bosque del Apache and Escondida subreaches, followed by the Elephant Butte and San Acacia subreaches, respectively. Overall, the San Acacia Reach showed greater habitat availability at low to moderate discharges relative to the Isleta Reach.

Time integrated habitat metrics (TIHMs) also differed between Reaches during the study period. TIHMs were nearly always larger in magnitude in the San Acacia Reach relative to the Isleta Reach (Figure 47a–c). 78 of the 81 TIHMs generated for the San Acacia Reach were greater than the respective TIHMs for the Isleta Reach (1993–2019) — of the TIHMs that were lower than the Isleta Reach, one was larval (1997) and two were adult (2002, 2016), however, the differences between these values were relatively small (range 0.3–3.9%). TIHM values in the San Acacia Reach were consistently greater than the Isleta Reach despite consistently lower discharges in the San Acacia Reach. Differences in hydrologic conditions and habitat availability between reaches illustrate the role geomorphology serves as a mediator between flow and habitat conditions in the Middle Rio Grande.

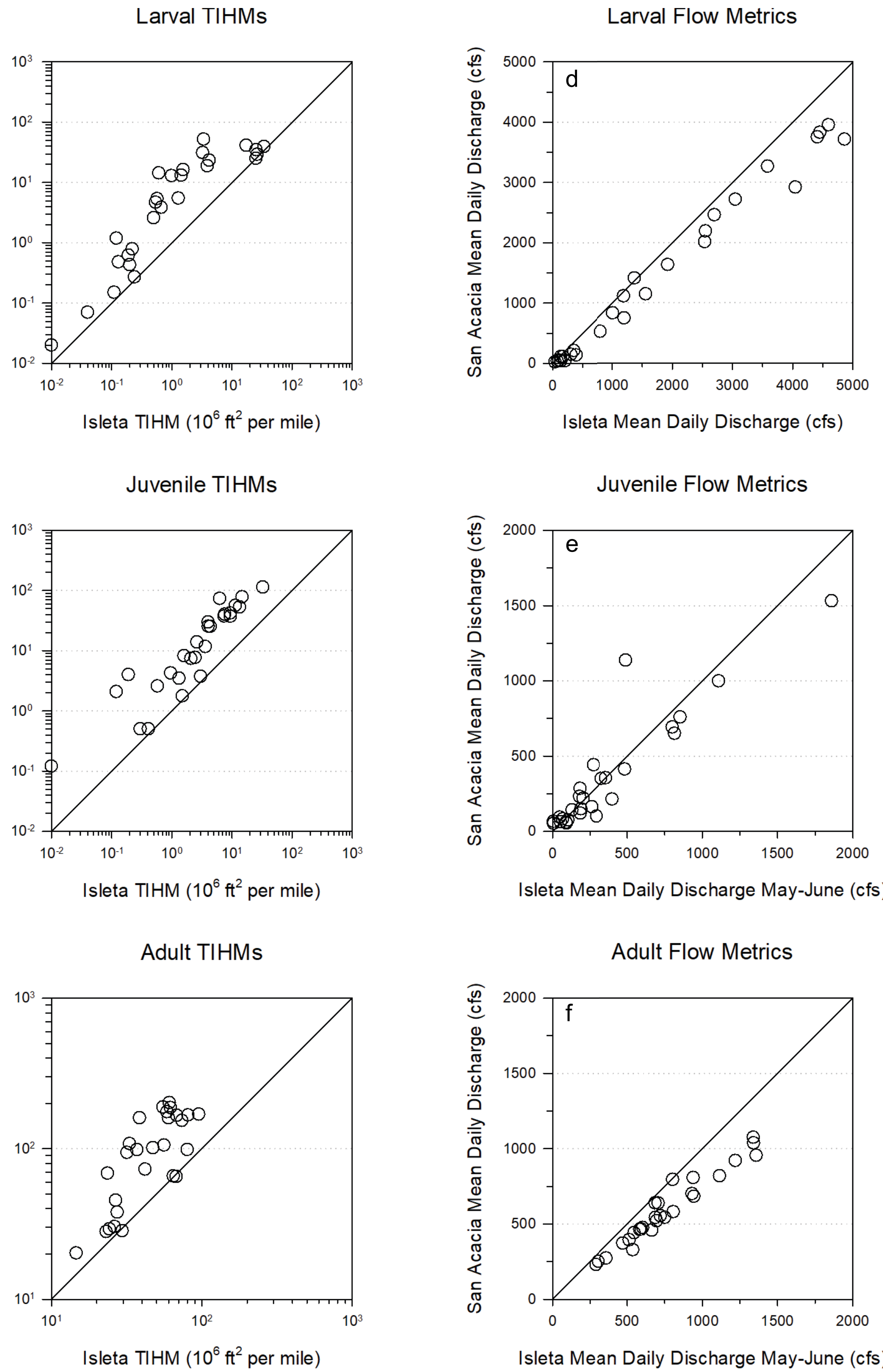



Figure 47. Comparison of Time Integrated Habitat Metrics (TIHMs; left) and flow metrics (mean daily discharge; right) between Isleta (x-axes) and San Acacia Reaches (y-axes).

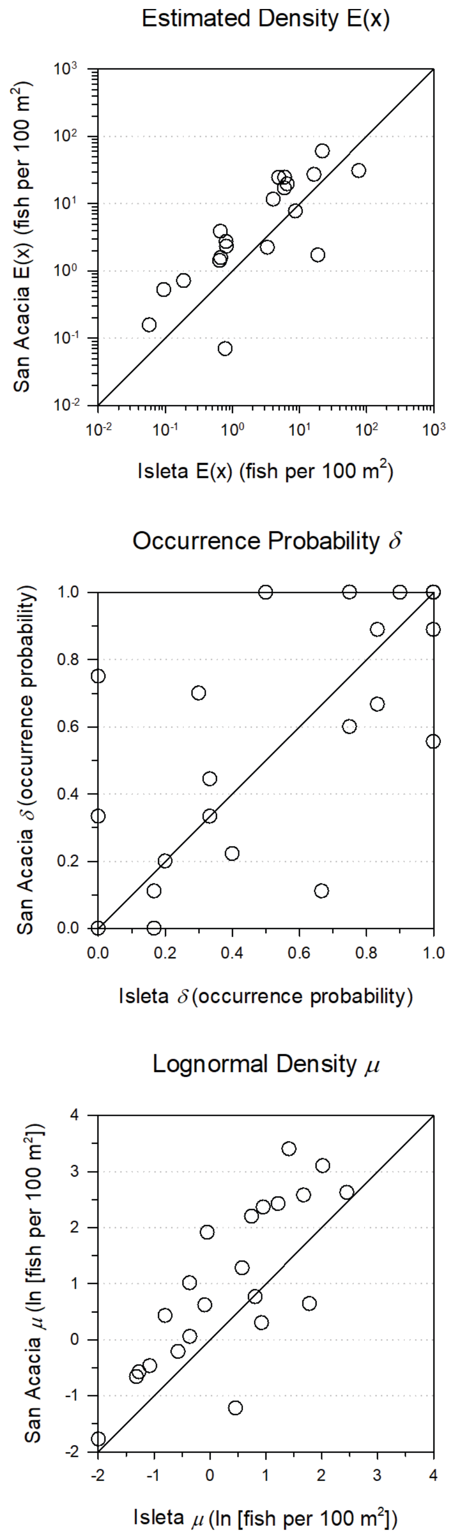



Figure 48. Comparison of estimated densities E($x$), occurrence probabilities ($\delta$), and lognormal densities ($\mu$) of the Rio Grande Silvery Minnow between Isleta (x-axes) and San Acacia Reaches (y-axes) 1993–2019.

### *Long-term Ecological Relationships*

Density and occurrence estimates of the Rio Grande Silvery Minnow at the reach-scale differed between the Isleta and San Acacia Reaches. Densities of the species were generally higher in the San Acacia Reach over time – no clear trend between reaches was observed for occurrence probabilities (Figure 48a–c). Although differences in density and occurrence estimates were evident between Reaches, these down-scaled values generally followed the same trends over time as those generated for the Middle Rio Grande (i.e., range-wide; Dudley et al., 2022).

Reach-scale analyses of relationships between the Rio Grande Silvery Minnow population and environmental variables produced relatively consistent results among the Isleta and San Acacia Reaches. Specifically, flow and habitat metrics corresponding to the larval life-stage (May–June) were the strongest predictors of positive population responses during the study period (Tables 8–10). For both Reaches, the highest ranked models for each modeling scenario (habitat and flow metrics separated and combined) were characterized by the same ecological metrics: larval TIHMs (MayJunHab; habitat metrics assessed independently) and prolonged, elevated flows May–June (MayJun28dHigh; flow metrics assessed independently and jointly). These results indicate the importance of seasonal hydrologic conditions for the Rio Grande Silvery Minnow across reaches of the species' current range.

Although the top ranked models for each modeling scenario were consistent between the Isleta and San Acacia Reaches, differences in subsequent model rankings were observed. For example, in the Isleta Reach, the top ecological models (flow and habitat metrics assessed together) were solely characterized by metrics corresponding to the larval life-stage (May–June) — in the San Acacia Reach, habitat metrics corresponding to juvenile and adult life-stages (i.e., JulSepHab, OctAprHab) were present in the top ecological models (Table 10). It is possible that factors represented by these habitat metrics were more consequential in the San Acacia Reach due to the higher incidence of low flows and flow intermittency in this Reach, however, the relatively low ranking of models containing these metrics warrants caution in the interpretation of these results. In both Reaches, flow metrics consistently outranked habitat metrics (all corresponding to the larval life-stage) in explaining interannual variation in species abundance and distribution. However, in the San Acacia Reach, larval habitat metrics received higher weighting than in the Isleta Reach. The reasons for improved performance in the San Acacia Reach are unclear yet hold promise for the development of habitat metrics undertaken in this study.

## *Key Process-Linkages in the Middle Rio Grande*

### *Process-Linkage 1: Floodplain Connectivity and Inundation*

The primary process-linkage identified in this study was floodplain connectivity and inundation. This linkage, previously identified in the Isleta Reach, shows the complex interactions among hydrologic and geomorphic processes in the Middle Rio Grande (Figure 49). 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 (Petrakis et al., 2017; Massong et al., 2006). In the San Acacia Reach, substantial lengths of the channel downstream of Escondida have progressed into the aggradational stages (Massong et al., 2010, 2006). This process has kept bankfull discharges relatively low (Figure 24), thereby maintaining connectivity between the main channel and floodplain. These channel characteristics contributed to elevated habitat availability in the San Acacia Reach relative to the Isleta Reach (as measured by TIHMs), notably for the larval life stage, despite consistently lower magnitude flows (Figure 47). Additionally, subreach contributions to larval TIHMs showed that Escondida and Bosque del Apache subreaches often contributed the majority of habitat at the reach-scale, further emphasizing the importance of floodplain connectivity to larval habitat availability (Figure 28). In contrast, the highly incised San Acacia subreach rarely contributed to larval habitat availability. The downstream-most subreach, Elephant Butte, also degraded 2002–2012 with similar impacts to larval habitat availability. The timing, duration, and frequency of floodplain 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 dependent on the seasonal availability of specific habitat conditions in the Middle Rio Grande 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). Floodplains decrease water velocities (i.e., increase hydraulic residence times), which reduce the downstream flux of eggs and larvae into unfavorable habitats, and increase the availability of shallow water depths, which provide spawning and nursery habitats for the Rio Grande Silvery Minnow (Valdez et al., 2019; Gonzales et al., 2014; Magaña 2012; Pease et al., 2006). The duration of nursery habitat availability 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 among the most reliable predictors of increased density and occurrence of the Rio Grande Silvery Minnow in the San Acacia Reach (Tables 8–10). Additionally, flow-habitat curves for larvae tended to show dramatic increases in habitat availability at bankfull discharge, indicating a strong relationship between larval habitat availability and floodplain inundation. In contrast to the Isleta Reach, which largely experienced reduced floodplain connectivity during the study period, the San Acacia Reach showed maintenance of floodplain connectivity, primarily in the Escondida and Bosque del Apache subreaches. 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 (Jacobson et al., 2014; Trinity River Restoration Program, 2009, The Nature Conservancy et al., 2008). 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.

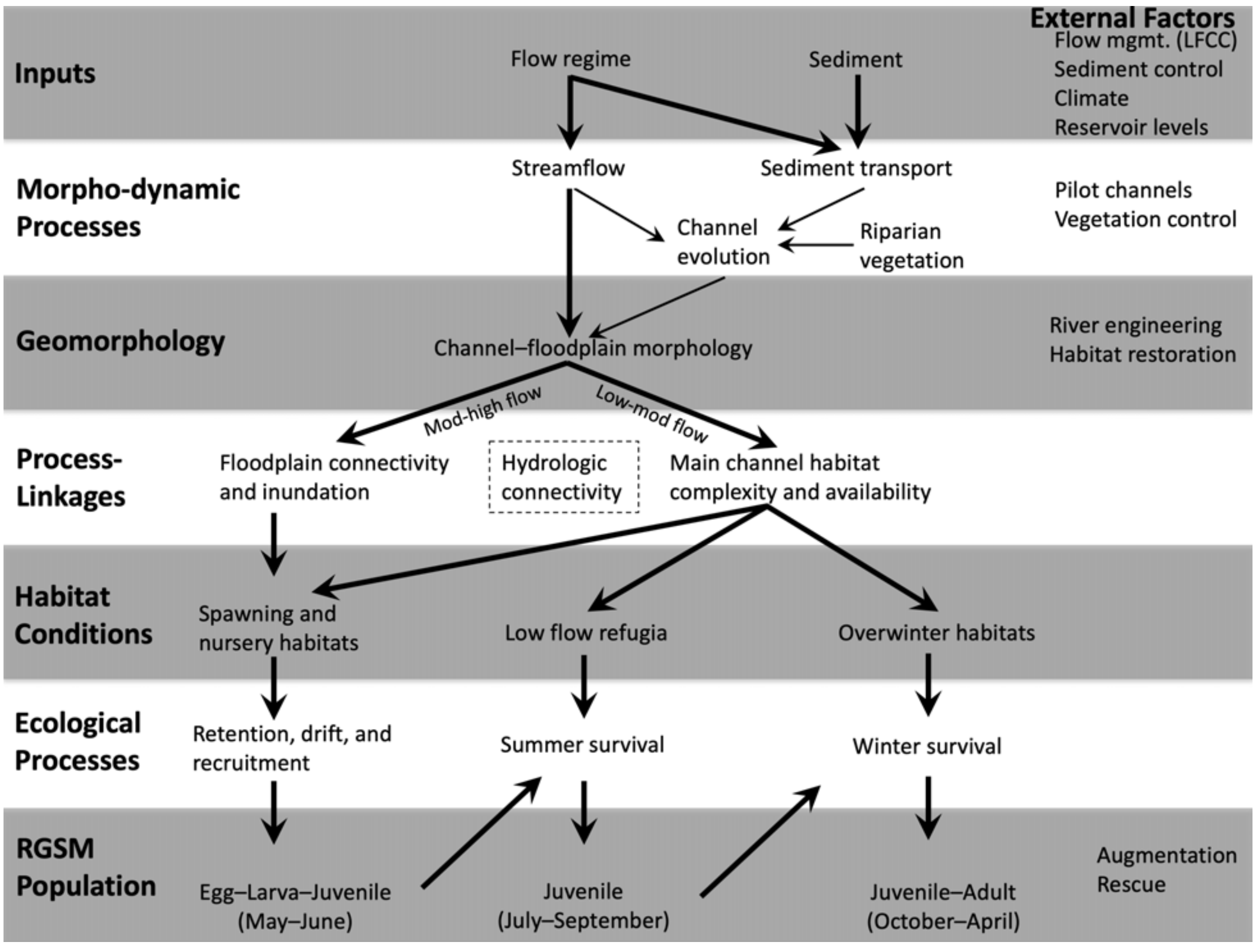



Figure 49. Conceptual diagram of key process-linkages in the Middle Rio Grande. Schematic based on the conceptual model of the Middle Rio Grande ecosystem (Figure 21). Arrows represent linkages among ecosystem processes and features. Line weights represent timescales; thicker arrows indicate linkages occurring over shorter timescales (i.e., seasonal, annual periods), thinner arrows indicate linkages occurring over longer timescales (i.e., multi-year periods).

*Process-Linkage 2: Hydrologic connectivity (within and among reaches)*

Hydrologic connectivity was identified as a secondary process-linkage in this study of the San Acacia Reach of the Middle Rio Grande (Figure 49). Here, hydrologic connectivity refers to the connectivity of lotic habitats within the San Acacia Reach (e.g., between subreaches) as well as the connectivity of the San Acacia Reach to upstream and downstream reaches (e.g., Angostura and Isleta Reaches, Elephant Butte Reservoir).

Sustained connectivity of lotic habitats within the San Acacia Reach is important to the survival of juvenile and adult Rio Grande Silvery Minnows. Prolonged low flow periods (<50–100 cfs) are typical in this reach during summer months, which can drastically reduce the availability and connectivity of lotic habitats when these periods cause flow intermittence in the reach (i.e., channel drying). Recent studies indicate that members of the Middle Rio Grande ichthyofaunal community, including the 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 and Reale, 2020; 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 hydrologic connectivity and the availability of lotic habitats within the reach should be prioritized to reduce mortality of the species during low flow periods.

The San Acacia Reach is the downstream-most reach of the Middle Rio Grande and is impacted by the suite of hydrological, morphological, and ecological conditions that occur upstream. In particular, the interaction of spawning activity by the Rio Grande Silvery Minnow and hydrological conditions in the Angostura and Isleta Reaches influences the magnitude of ichthyofaunal drift (i.e., downstream dispersal of eggs and larvae) into the San Acacia Reach. 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 into the San Acacia Reach will be increased. Downstream reaches (i.e., Isleta and San Acacia Reaches) have nearly always contained 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 and larvae (Dudley et al., 2022). However, this study showed high habitat availability in the San Acacia Reach relative to the Isleta Reach, which likely also contributes to higher densities in the San Acacia Reach. Altogether, population responses in the San Acacia Reach are not entirely representative of localized habitat conditions within the reach, rather, such responses are also influenced by complex interactions of factors in upstream reaches and their impacts on the dispersal of ichthyofauna throughout the Middle Rio Grande.

The downstream dispersal of eggs and larvae of the Rio Grande Silvery Minnow carries additional implications for this species in the Middle Rio Grande. The maintenance of floodplain connectivity and habitat availability in the San Acacia Reach suggests more favorable habitat conditions for the retention and survival of early life-stages of the species relative to upstream reaches (e.g., Isleta Reach). However, as the downstream-most reach with restricted upstream dispersal opportunities due to San Acacia Diversion Dam, it is unlikely that this reach, in isolation, could provide long-term population stability given life history requirements of the species (Mortensen et a., 2019; Platania et al., 2020). The population of the Rio Grande Silvery Minnow in the San Acacia Reach is likely dependent on upstream spawning activity, and over time, net downstream dispersal of offspring would likely result in progressive upstream losses given the absence of upstream dispersal, augmentation, or relocation efforts to redistribute juveniles and adults upstream. Additionally, heightened frequency of flow intermittence in this reach contributes to elevated mortality rates that are not experienced upstream (e.g., Angostura Reach). Cumulatively, these factors demonstrate the importance of maintaining hydrologic connectivity within and among the reaches of this river system for this species.

Downstream hydrologic connectivity between the San Acacia Reach and Elephant Butte Reservoir also revealed impacts to morphological processes proximal to the reservoir. Reservoir levels (i.e., pool elevation) influence sediment transport in downstream subreaches (e.g., Elephant Butte subreach; Holste, 2015). Reservoir levels varied substantially during the study period (1962–2012; Figure

39) with coincident impacts to morpho-dynamics during substantial shifts in reservoir levels. Higher reservoir levels coincided with periods of increased channel aggradation in the Elephant Butte subreach (1972–2002); conversely, lower reservoir levels coincided with periods of channel degradation (2002–2012). In addition to morphological effects, reservoir levels control the distribution of lotic (i.e., flowing water) and lentic habitats (i.e., standing water) near the reservoir – Rio Grande Silvery Minnow require lotic habitats. While it is unlikely that these hydrologic and geomorphic processes have considerably impacted the reach-scale population of Rio Grande Silvery Minnow due to the localized nature of the impacts and their location at the downstream-most end of the species' range, the occurrence of these processes further illustrates the role hydrologic connectivity serves in the Middle Rio Grande.

*Process-Linkage 3: Main Channel Habitat Complexity and Availability*

Main channel habitat complexity and availability was identified as a secondary process-linkage in this study (Figure 49). Similar to floodplain connectivity, 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.

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 (Figure 49). 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 the availability of lotic habitats or low flow refugia to survive harsh environmental conditions that can occur during low flow periods (e.g., high water temperatures, low dissolved oxygen). During October–April, juveniles and adults depend on the availability and stability of overwinter habitats to survive until spring. During the winter, this species is more commonly associated with areas of high habitat complexity, notably debris piles and deeper, low-velocity habitats (Dudley and Platania, 1997). This study considered how changes to instream habitat complexity and availability impact the population of the Rio Grande Silvery Minnow.

Geomorphic trends in the San Acacia Reach carry similar implications for main channel habitat complexity and availability that were noted in the Isleta Reach. Generally, reduced channel width has contributed to a reduced main channel habitat complexity and availability, particularly the loss of shallow, low-velocity habitats within the channel. Changes to main channel complexity can be attributed to factors including channelization, flow regulation, and sediment control. In contrast to the Isleta Reach, which showed channel narrowing and incision, channel aggradation in the San Acacia Reach (e.g., E, BDA, EB subreaches) might contribute to the maintenance of habitat complexity within the contemporary channel, however, difficulties assessing habitat conditions at low flows in this study limited inferences regarding changes to main channel habitat availability over time. Effects of geomorphic changes on habitat metrics showed reductions to habitat availability over time at low to moderate flow, suggesting this reach has also experience considerable changes to main channel habitat complexity and availability. Despite the potential for differences in sediment transport processes to impact main channel habitat complexity in the San Acacia Reach, maintaining flowing conditions and therefore the availability of lotic habitats is likely to be more consequential to the Rio Grande Silvery Minnow.

Although main channel complexity has been impacted in the San Acacia Reach, 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 flow and habitat metrics corresponding to juvenile and adult life-stage periods were relatively weak predictors of abundance and occurrence of the Rio Grande Silvery Minnow in the San Acacia Reach (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 (Linkage 1), 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 moderate discharges (e.g., 500–3,000 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 instream bars and islands to substitute for floodplain connectivity in incised river channels (McComas et al., 2022). Overall, our findings suggest that maintaining habitat availability through instream flows, particularly during seasonal low flow periods (July–September), is anticipated to produce positive population responses.

*Long-Term Ecological Relationships*

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 flow and habitat metrics indicated that flow metrics consistently explained more variation, as compared to habitat metrics, across years (1993–2021). Habitat metrics might not have fully captured spatial and temporal variations of floodplain inundation and habitat formation. This factor was also noted for the Isleta Reach where flow-habitat relationships showed sharp increases in habitat availability over a small range of discharges (corresponding to exceedance of bankfull discharge), thus, small differences in flows often had notable impacts on the predicted amount of larval habitat – these modeled responses might 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 multiple years. In comparison, the San Acacia Reach showed lower bankfull discharges and more gradual increases in habitat availability to increases in discharge largely due to the presence of aggrading subreaches downstream (e.g., BDA and EB subreaches). Yet, perched channel conditions in the San Acacia presented modeling challenges that may have contributed to inaccuracies in modeling inundation of low-lying areas. These factors might partially explain why flow metrics, as compared to habitat metrics, were consistently more predictive of the increased occurrence and density of this species over time.

In both the Isleta and San Acacia Reaches, the highest-ranking flow metrics corresponded to peak discharge over a 28-day period during May–June. 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 in relation 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. While these metrics were not chosen to provide detailed assessments across sites or sub-reaches, 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. 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). Interestingly, habitat metrics corresponding to juvenile and adult life-stages were present in the top ecological models for the San Acacia Reach (all metrics combined; Table 10), whereas these metrics were absent from Isleta Reach analyses. Accordingly, habitat conditions for juveniles and adults should not be entirely neglected as adverse conditions caused by extremely low flows 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).

*Geomorphic Controls on Habitat Conditions*

The key process-linkages identified for the San Acacia Reach illustrate the dynamic and complex interactions among streamflow, channel-floodplain morphology, and hydraulic habitat conditions. The San Acacia Reach showed varied channel evolution trends during the study period. This progression is evident through the planform evolution model for the Middle Rio Grande (Massong et al., 2010) – herein referred to as the MRG model. Over time, subreaches in the San Acacia Reach were characterized by the three stage classifications described by the MRG model: initial (1–3), aggrading reach (A4–A6), and migrating reach (M4–M8). Overall, the Reach showed a gradual transition over time from the initial stages to migrating reach stages in the upstream-most subreaches (SA1–E1) to aggrading reach stages further downstream (subreaches E4–BDA5). In terms of geomorphic processes, these stages indicate a shift from excessive to deficient sediment transport capacity (Figures 11 and 34). The Isleta Reach, which transitioned exclusively to migrating reach stages, was characterized by widespread channel incision and floodplain disconnection attributed to excessive sediment transport capacity. In contrast, the transition to aggrading reach stages in the San Acacia Reach was associated with maintained connectivity to the floodplain (i.e., reduced bankfull discharge) and relatively high habitat availability at moderate–high flows.

Planform evolution, as described by the MRG model, contains processes not fully observed in the San Acacia Reach. In the upstream subreaches (SA1–E1), the channel tended to progress toward the migrating stages of the MRG model, however, progression to the latter migrating reach stages (M6–M8) were not observed. The absence of these model stages could be attributed to two factors, (1) channel incision has not been great enough to undermine banks and cause bank erosion and/or (2) bank erosion is restricted by bank armoring (e.g., jetty jacks and riparian vegetation). The first factor is controlled by 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. These 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 incision and widening (Booth and Fischenich, 2015; Smith et al., 2008). 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 (Johnson et al., 2020; Castro and Thorne, 2019; Booth and Fischenich, 2015; Cluer and Thorne, 2013). Additionally, in downstream subreaches (E4–BDA5), the channel progressed toward the aggrading reach stages of the MRG model – these stages describe the potential for sediment plug formation (stage A5; observed), which causes the flow to develop a new channel, ultimately causing a channel avulsion (stage A6; not observed). While multiple sediment plugs have formed in the Bosque del Apache and Elephant Butte subreaches since 1990, in each of these instances a pilot channel was dug through the sediment plug thereby returning flow to the original channel alignment and preventing lateral channel migration to occur naturally (Massong et al., 2010). 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). The complex nature of sediment plug formation and channel avulsion processes are difficult to predict and control yet might reveal an opportunity to harness natural processes to improve habitat conditions in this reach given the right conditions. The severe degree of channel perching that occurs in locations where recent sediment plugs have tended to form, might cause undesirable effects if channel avulsions were allowed to occur naturally (e.g., uncontrolled flooding, threaten levees/infrastructure). Channel avulsions occurred relatively frequently in the Rio Grande historically and these events helped maintain channel complexity over time, however, it is unclear how changes to the channel (e.g., channelization/jetty-jacks, dense riparian vegetation) might affect such processes. Insights gained from the synthesis of channel evolution models and habitat analyses might be used to inform habitat restoration efforts of the Middle Rio Grande.

### *Analytical Considerations*

#### *1D Hydraulic Modeling (HEC-RAS)*

Several limitations of 1D modeling were identified in this study. These limitations, which were noted during the Isleta Reach analyses, were primarily associated with estimating bankfull discharges, channel geometry resolution, and lateral flow distributions. Despite limitations identified with the applied modeling approaches, this study utilized and integrated long-term datasets of the Middle Rio Grande to the extent feasible given the scope of research.

Estimating bankfull discharges was challenging given data and modeling constraints. HEC-RAS top of bank points were placed to contain the flow within the channel until the estimated bankfull discharge. At discharges equal to or exceeding the estimated bankfull discharge, the top of bank points were removed to allow water to flow onto the floodplains. Cross-section geometry, and therefore bankfull discharge, varies within and throughout reaches, which makes accurate estimation of bankfull discharges difficult. This issue was especially prominent in the San Acacia Reach due to substantial lengths of the river subject to channel perching conditions (e.g., Escondida and Bosque del Apache subreaches). The use of top of bank points to constrain the lateral distribution of water in the model likely resulted in conservative estimations of inundated areas; this effect was also noted in the Isleta Reach. This issue was mitigated through consultation with USBR and by setting a modeling threshold (25% of cross-section overtopped) to estimate bankfull discharge values. This approach was systematically applied to both Isleta and Acacia Reach analyses, providing a basis for comparing reach-scale results. It is important to note that the estimation of bankfull discharges strongly influenced the shape of flow-habitat curves and TIHM calculations (see below). Ground truthing or aerial surveys would be needed to improve accuracy and confidence of estimated bankfull discharges, however, such observations would only pertain to current channel conditions and would be limited to flows present during surveying.

Another limitation of 1D modeling involved the resolution of channel geometry data. Cross-section geometry data were obtained at agg/deg lines, which are spaced approximately every 500 ft. Because HEC-RAS uses the cross-section geometry data at the agg/deg lines, the lateral flow distribution could only be obtained at 500 ft intervals because the 1D model used does not account for variable flow distributions between cross-sections (Baird and Holste, 2020). Therefore, the width of available habitat at a cross-section was assumed to remain constant between cross-sections. Although this assumption is physically unrealistic, it provided a means to estimate habitat availability at relatively large spatial scales (e.g., reach-scale). Overall, it was accepted that 1D modeling might not provide highly accurate estimates of habitat availability, rather, our estimates of habitat availability were considered to provide suitable metrics for assessing spatial and temporal variation in habitat availability in the Middle Rio Grande and their relationship to the population of the Rio Grande Silvery Minnow.

The resolution of channel geometry also limited modeling of flow-habitat relationships at low flows (<500 cfs). Channel elevations beneath the water surface during channel surveys are approximated by an idealized trapezoidal cross-section. Thus, for modeling discharges below the discharge at the time of survey, flow is contained entirely within a trapezoidal cross-section, which likely does not accurately represent the availability of hydraulically suitable habitats in the main channel habitat at these flows. This study focused on the range of discharges expected to produce reasonable hydraulic modeling results (e.g., 500–10,000 cfs), however, low flow periods occurred frequently during summer months (July–September; occasionally in October) in the San Acacia Reach, which affected TIHM calculations for these periods. Low flow habitat relationships have been previously investigated in the Middle Rio Grande using 2D modeling techniques and high-resolution, site-scale elevation data (Bovee et al., 2008). That study indicated a relatively sharp peak in availability of hydraulically suitable habitats (i.e., using velocity and depth criteria) at low flows of about 100–200 cfs. However, it is important to consider that additional factors beyond hydraulics (e.g., water quality, predation, competition) likely become more consequential during low-flow periods and the potential for flow intermittence also increases at such low discharges.

### *Data Availability*

This study analyzed several systematically collected long-term datasets of the Middle Rio Grande. These datasets were collected at different time intervals and over different periods, which affected our ability to fully integrate them. Accordingly, this study used an annual time step for the period 1993–2021, which was consistent with available ecological and hydrological datasets. While the analyses performed herein were reasonable given currently available datasets, several relevant considerations related to data availability are important to acknowledge.

For the geomorphic datasets, channel geometries were obtained at approximately 10-year intervals between 1962 and 2012, excluding 1982. In a dynamic, mobile-bedded river such as the Rio Grande, the channel could change considerably between surveys, thereby reducing the accuracy of hydraulic modeling results for years between surveys. However, in the San Acacia Reach, the largest magnitude of channel change (e.g., channel incision and narrowing) occurred between 1972 and 1992 — recently (i.e., 1992–2012), this reach does not appear to have consistently experienced drastic year-to-year changes. Thus, for the Isleta and San Acacia Reaches, the 10-year intervals between surveys appeared to be sufficient for characterizing recent temporal variation in channel and habitat conditions at the spatial scales investigated in this study. Increased frequency of channel surveys (e.g., 5-year increments) would help verify the magnitude of temporal change and its impact on hydraulic habitat assessments. Also, because channel geometry is not available after 2012, flow-habitat curves were unable to be interpolated for the period 2013–2021. Therefore, the analysis of these years does not account for the most recent channel changes and their potential effects on habitat conditions. Given the relatively low magnitude change in flow-habitat curves 1992–2012, it is not expected that the period 2013–2021 would substantially differ from 2012.

For the hydrologic datasets, increased data availability improved spatial characterization of flow conditions in the San Acacia Reach relative to the Isleta Reach. For the Isleta Reach, multiple gaging stations were used to create a single hydrograph for the reach due to missing periods of record among gaging stations. For the San Acacia Reach, multiple gaging stations provided means to characterize flow conditions for each of the four subreaches investigated. Data for intermediate gaging stations (USGS 08355050 Rio Grande at bridge near Escondida, NM; USGS 08355490 Rio Grande above US HWY380 near San Antonio, NM) were not available until WY 2006, however, gages at the upstream and downstream ends of the study area were available for 1993–2005 (USGS 08354900 Rio Grande floodway at San Acacia, NM; USGS 08358400 Rio Grande floodway at San Marcial, NM). The increased availability of hydrologic data in the reach improved characterization of TIHMs by accounting for spatial variation in flow conditions within the reach. As observed in the Isleta Reach analyses, low flows and flow intermittence remained a challenging hydrologic condition to account for spatially and temporally. Improved characterization and understanding of discharge-drying relationships could improve our confidence in assessing habitat availability and calculating TIHMs during low flow periods.

Collection of LiDAR data and aerial photography at multiple temporal scales could also be useful for understanding interactions between streamflow and channel morphology. For example, seasonal data collection could be coordinated to capture biologically-relevant hydrologic periods and help characterize within-year variation of habitat conditions during these periods (e.g., spring – peak flow, summer – low flow, autumn/winter – steady flow). Potentially, this data could be used to improve accuracy of bankfull discharges by subreach, thereby improving accuracy of flow-habitat relationships and subsequent calculations of habitat availability metrics (TIHMs). Additionally, LiDAR data could be collected during low flow periods to obtain channel surface elevations (i.e., reduce area of channel estimated by an idealized trapezoidal cross-section), thereby increasing modeling capabilities and accuracy at low discharges ($Q<500$ cfs). Aerial photography obtained during river drying episodes could also be used to better characterize relationships between discharge and drying extent.

### *Flow-Habitat Curves and Time Integrated Habitat Metrics*

A key consideration for the evaluation of flow-habitat curves and Time Integrated Habitat Metrics (TIHMs) involved the estimation of bankfull discharges. Specifically, TIHMs were particularly sensitive to the inflection points of the flow-habitat curves. Inflection points are shown by sharp increases in habitat availability over a relatively small discharge increment (about 500 cfs) representing the estimated bankfull discharge and occurrence of floodplain inundation. While it was suspected that habitat availability might have been underestimated for certain discharges in the Isleta Reach (i.e., flows just below estimated bankfull discharge), the accuracy of estimated bankfull discharges in the San Acacia Reach is unclear (Mortensen et al., 2020). Given current data availability and modeling approaches (e.g., coarse cross-sectional data, 1D hydraulic modeling), improving accuracy of bankfull discharges is likely to remain an analytical consideration. Ongoing and future data collection efforts should seek to address this limitation.

Accuracy of flow-habitat curves and subsequently, TIHMs, were also uncertain at low flows. Modeling accuracy at low flows (<500 cfs) was limited by cross-sectional data – channel elevations below the water surface at the time of data collection (generally >500 cfs) were typically estimated using an idealized trapezoid method. This simplification of the channel limited the capability of the model to estimate hydraulically suitable habitat areas within the channel at low flows. In the San Acacia Reach, low flows are particularly common especially during summer months. As such, juvenile and adult TIHMs were likely the most affected — these metrics were found to be less reliable predictors of the occurrence and distribution of the Rio Grande Silvery Minnow as compared to peak flow periods (i.e., larval life-stage May–June), however, the absence of accurate channel elevations for low flow conditions might have reduced reliability of these metrics in our analyses. Despite this consideration, greater availability of streamflow data in the San Acacia Reach helped improve characterization of spatially variable hydrologic conditions for the generation of habitat metrics in this Reach.

### *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 Isleta Reach because it has incised during the study period (Mortensen et al., 2020). Therefore, it might be more accurate to add designations to stages 2 and 3 (e.g., M2 and M3) to better describe this degradational trend. Additionally, the roles of external factors were not incorporated into the MRG model. This study noted that reservoir levels impacted channel evolution processes in the downstream-most subreach (Holste, 2015) – 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 have not progressed to the point of developing a new channel alignment naturally (stage A6), rather pilot channels have been 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 surveys of channel morphology will improve our understanding of ongoing geomorphic processes and resulting evolutionary trajectories for the Middle Rio Grande.

*Long-Term Ecological Relationships*

This study identified several limitations associated with analyses of ecological relationships within reaches of the Middle Rio Grande (i.e., the population of the Rio Grande Silvery Minnow). These limitations included downscaling population metrics and sensitivity to select environmental variables.

The Rio Grande Silvery Minnow Population Monitoring Program has systematically provided annual assessments of occurrence and abundance of the species for the Middle Rio Grande (i.e., range-wide). For the purposes of our study, population metrics were downscaled to the reach-scale. Although it is possible to downscale density estimates, these results are subject to limitations (e.g., decreased sampling size, increased variance, increased confidence intervals). In particular, the decreased sample size at smaller spatial scales results in data gaps. The increased data gaps at the reach scale, and particularly at the subreach scale, are concerning because low numbers of samples could potentially lead to incomplete or spurious inferences regarding long-term ecological relationships with the Rio Grande Silvery Minnow (e.g., flow-RGSM or habitat-RGSM relationships). This problematic effect is notably increased as the number of sites is reduced with downscaling, resulting in fewer years available with estimated density and occurrence parameters (e.g., during drought periods). Based on relatively poor data reliability for the downscaled estimates, we exercised caution in using these reach results for analyses of available population monitoring data. The San Acacia Reach, as compared to the Isleta Reach, contained a greater number of sampling sites that were monitored more frequently during the study period, thereby improving confidence and reliability of population estimates at the reach-scale. Population trends were remarkably similar between Reaches and range-wide estimates (i.e., Middle Rio Grande), suggesting that downscaling (of either Reach) did not prohibitively affect population estimates at the reach-scale. Also, analyses of ecological relationships identified the same specific flow and habitat conditions in both the Isleta and San Acacia Reaches, thereby confirming previous results and further emphasizing the importance of these factors across the species' range. Given that the Rio Grande Silvery Minnow Population Monitoring Program was designed to assess range-wide variation in the occurrence and density of the species, a full integration of spatial scales (i.e., from discrete reaches to the species' range) would likely provide further insight into population dynamics of this species within the Middle Rio Grande.

Analyses of long-term ecological relationships indicated challenges associated with metrics of habitat availability. In particular, flow-habitat curves, and subsequently habitat metrics (TIHMs), were highly sensitive to estimated bankfull discharge values. This effect, which was also noted in the Isleta Reach, might have impacted our ability to elucidate strong relationships between habitat metrics 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 constraints and methods. 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 habitat metrics by life-stage across years and reaches (i.e., Isleta versus San Acacia Reach). In addition to the flow and habitat 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.

### *Potential 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 the San Acacia Reach habitat analyses carry potential implications for the restoration of aquatic habitats in the Middle Rio Grande.

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). Channel aggradation in the San Acacia Reach appeared to maintain floodplain connectivity at moderate to high flows in this study. The maintenance of floodplain connectivity via natural processes in this reach suggests that restoring floodplain connectivity should prioritize upstream reaches that have experienced reductions or loss to floodplain connectivity (e.g., Isleta Reach, upper San Acacia Reach). 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 is expected to be beneficial because such actions might contribute to higher rates of propagule retention in upstream reaches. Although the San Acacia Reach appears to have more favorable habitat conditions relative to the Isleta Reach, the downstream location of this reach within the species' range increases the likelihood that eggs/larvae produced within this reach will drift into Elephant Butte Reservoir, an unsuitable habitat for recruitment and survival of this species. Accounting for location and local channel processes indicates certain subreaches are likely to be unfavorable targets for restoration. For example, the Elephant Butte subreach is the downstream-most subreach and channel evolution in this subreach is influenced by downstream reservoir levels. Therefore, its downstream location and susceptibility to unpredictable, external factors suggest this location is not an ideal candidate for habitat restoration. Nonetheless, habitat conditions within the San Acacia Reach could also be improved, however, any restoration activities in this area must consider the morpho-dynamic trends present within this reach (e.g., high rates of aggradation, perched channels, lower flows).

Our understanding of habitat restoration projects is currently limited with regard 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 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 (Valdez et al., 2019; Gonzales et al., 2014), however, it is unclear how these restored habitats contribute to population dynamics beyond the site-scale (e.g., reach-scale, range-wide). Such assessments are 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 to the Isleta and San Acacia Reaches 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 (The Nature Conservancy et al., 2008; Stillwater Sciences, 2007). Additional research efforts, environmental organizations, and river managers support these recommendations (Florsheim et al., 2008; Smith et al., 2008; Olson et al., 2014). 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 et al., 2009; Jacobson and Galat, 2006). 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 (Yarnell et al., 2015; 2010; Arthington et al., 2006). 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 in the Isleta and San Acacia Reaches and the Rio Grande Silvery Minnow population over time. The strongest ecological relationships evaluated were between increased magnitude and duration of spring flows, increased availability of shallow, low-velocity habitats (i.e., larval habitats), 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 are relatively clear – overbank flows need to recur at a frequency that provides successful recruitment of the Rio Grande Silvery Minnow and 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). Given the relatively low overbanking discharges downstream of Escondida observed during this study, the San Acacia Reach presents a potential opportunity to drastically increase larval habitat availability in this reach through lower magnitude increases in discharge relative to upstream reaches (e.g., Isleta Reach). 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).

While spring runoff and habitat availability during this period were shown to be the strongest predictors of the abundance and occurrence of the Rio Grande Silvery Minnow in autumn, base flow conditions during summer, autumn, and winter influence survival rates during these periods (Linkage 3). When base flows are extremely low, habitat availability and quality can rapidly deteriorate as lengths of river become intermittent and habitats become restricted to isolated pools, causing high rates of mortality (Archdeacon and Reale, 2020; Van Horn et al., 2022). Considerable lengths of the Isleta and San Acacia Reaches run dry on a near-annual basis, limiting survival of juveniles and adults, and therefore, reducing or preventing intermittency should remain a priority. In combination with providing adequate spring flow conditions, managing base flows, particularly during summer when extremely low flows are most likely to occur, is expected to produce positive population responses over time (Hatch et al., 2020). Although the flow management implications of spring and summer flows to the population of the Rio Grande Silvery Minnow are evident, 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 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 Isleta and upper San Acacia Reaches 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 largely focused on evaluating spatiotemporal patterns of the physical 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; USOFR 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 San Acacia 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-dynamics processes and habitats needed by the Rio Grande Silvery Minnow.

The main findings of this study included:

- Key process-linkages identified for the San Acacia Reach 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 San Acacia Reach showed distinct, spatially variable trends over time (1962–2012) that differed considerably from the Isleta Reach (Linkage Report I).
- Discharge was consistently lower in the San Acacia Reach compared to the Isleta Reach (including increased frequency of intermittency), however, habitat metrics were consistently greater during the study period (1993–2021).
- Channel aggradation was prevalent downstream of Escondida, which corresponded to floodplain connectivity and greater larval habitat availability – water surface elevation at Elephant Butte Reservoir was shown to control morpho-dynamics in the downstream-most subreaches over time.
- Densities of the Rio Grande Silvery Minnow were generally higher in the San Acacia Reach relative to the Isleta Reach. Higher densities were attributed to greater larval habitat availability and pertinent ecological processes (i.e., downstream drift/dispersal).
- Flow and habitat metrics corresponding to the larval life-stage of Rio Grande Silvery Minnow were the most reliable long-term predictors of the species' density and occurrence at the reach-scale, however, flow metrics explained more variation in population parameters across years.
- Habitat metrics showed that the greatest increases in larval habitat availability (i.e., several orders of magnitude) were linked to prolonged overbanking flows.
- 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. Current limitations to hydraulic modeling and habitat analyses include the estimation of overbanking discharges for perched/semi-perched channels and limited accuracy of modeling low flows.
- Flow management in the San Acacia Reach will be important to the recovery of the Rio Grande Silvery Minnow. Relatively low overbanking discharges and floodplain connectivity downstream of Escondida suggest abundant larval habitats can be created given sufficient spring runoff, however, high frequency of intermittency during the summer is detrimental to survival. Restoration of larval habitats is expected to be most effective between San Acacia Diversion Dam and Escondida due to high channel incision, perennial flows, and upstream location.

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 biology, ecology, engineering, hydrology, and geomorphology contributed to valuable insights into the complex dynamics influencing habitat conditions needed by this imperiled species. This was the second Linkage Report produced for this project, which included an assessment of process-linkages for the San Acacia Reach of the Middle Rio Grande (San Acacia Diversion Dam to San Marcial, NM). Results from this reach were compared to the Isleta Reach (Linkage Report I). Future Linkage Reports will incorporate additional reach analyses as they become available (i.e., Angostura Reach). Continued progress on the recovery of the Rio Grande Silvery Minnow will depend on the ongoing support of river managers to actively pursue research and monitoring efforts that inform management of flows, aquatic habitats, and ecological resources.

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## KEY TERMS AND DEFINITIONS

This 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.

*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 term used to quantify the approximate volume of water lost during storage and delivery of surface water resources.

*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 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 for the purposes of this report, this term 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 the Rio Grande Silvery Minnow. Relationships between discharge and habitat availability (i.e., flow-habitat curves) were obtained by 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 a fish.

*Morphology (fluvial)* describes the form and structure of a stream 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 (e.g., Angostura and Isleta Reaches) and study area boundaries (e.g., San Acacia Reach).

*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 of the watershed.

*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 habitat metric proposed for this study that relates the timing and persistence of hydraulically suitable habitats to the life history of the Rio Grande Silvery Minnow. TIHMs represent the integral of habitat availability over time for specific periods corresponding to the principal life-stages (e.g., larvae, juvenile) of the Rio Grande Silvery Minnow.

# APPENDIX A
# SUPPLEMENTARY RESULTS

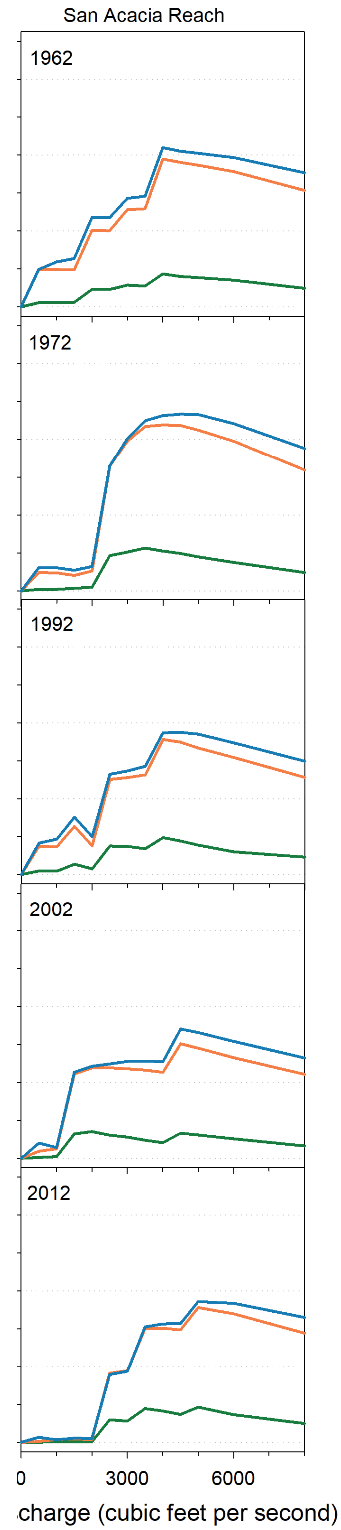
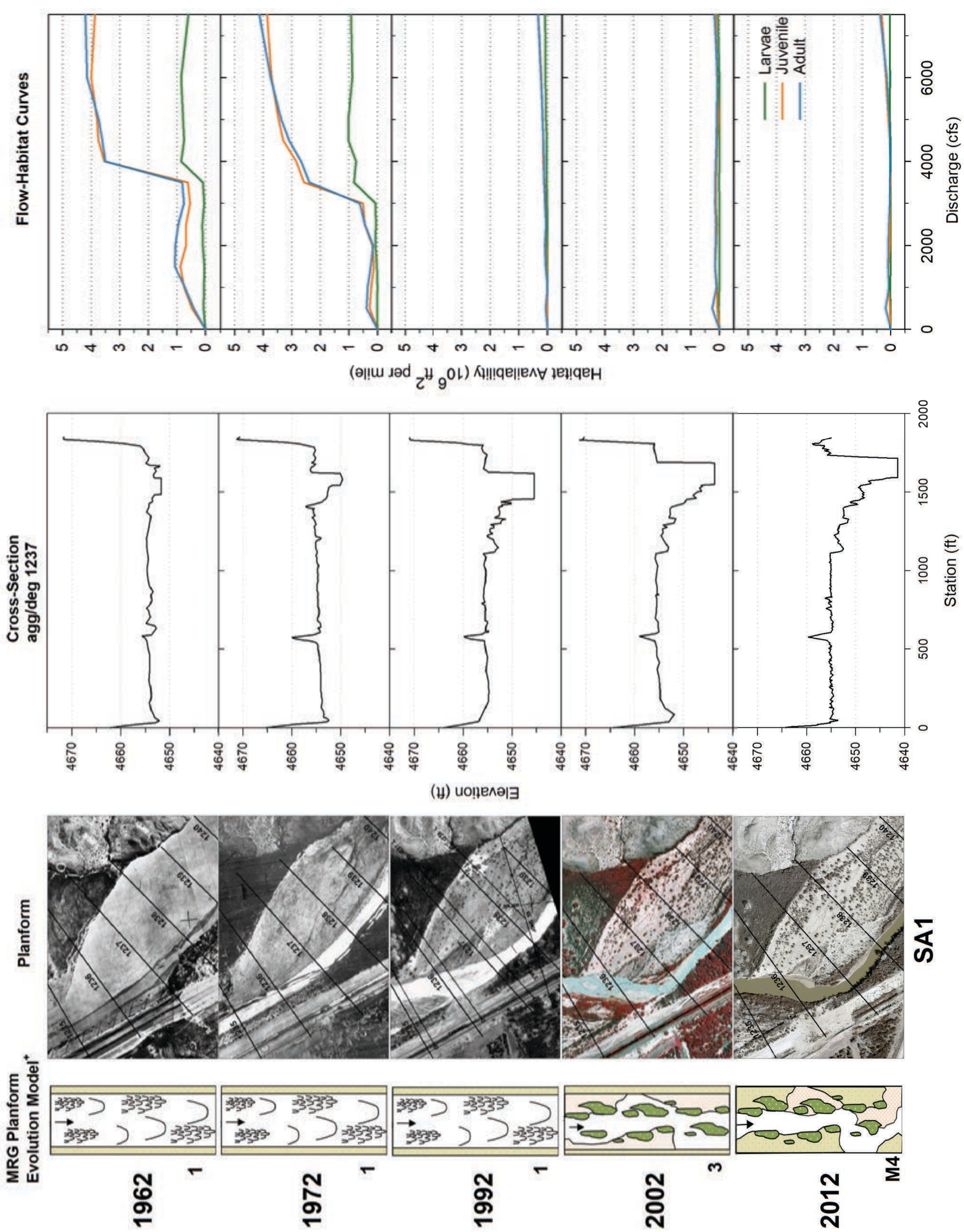
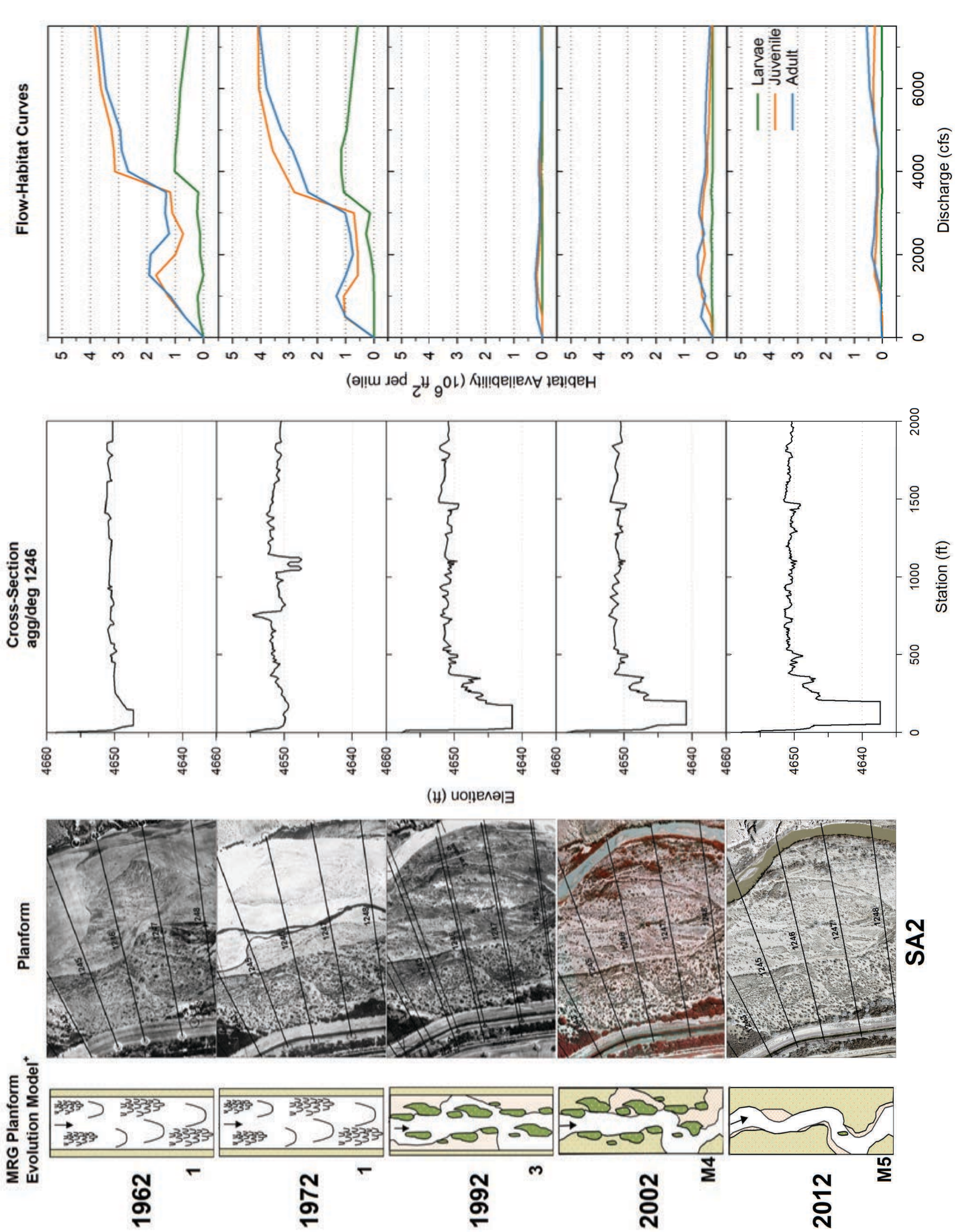
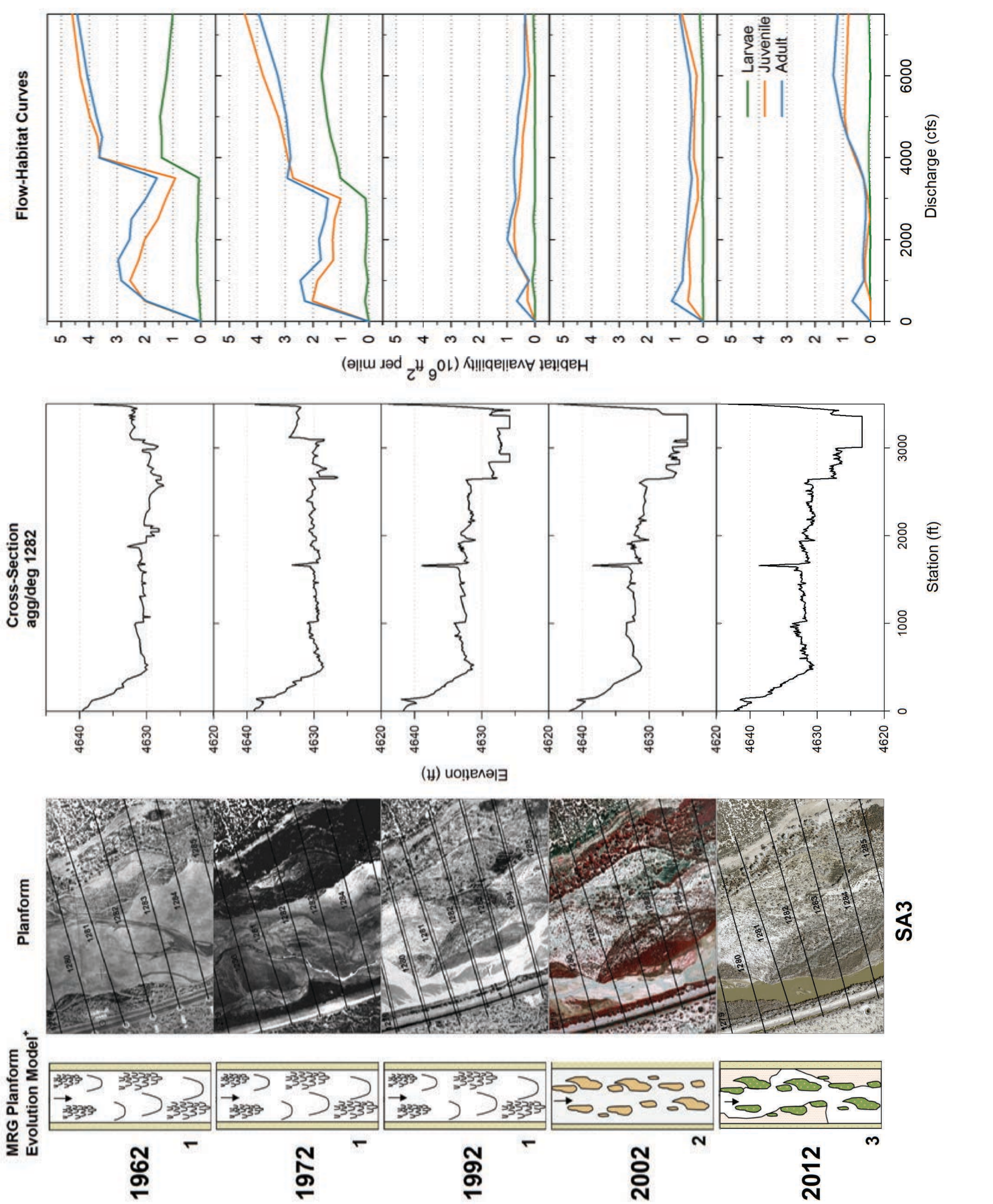
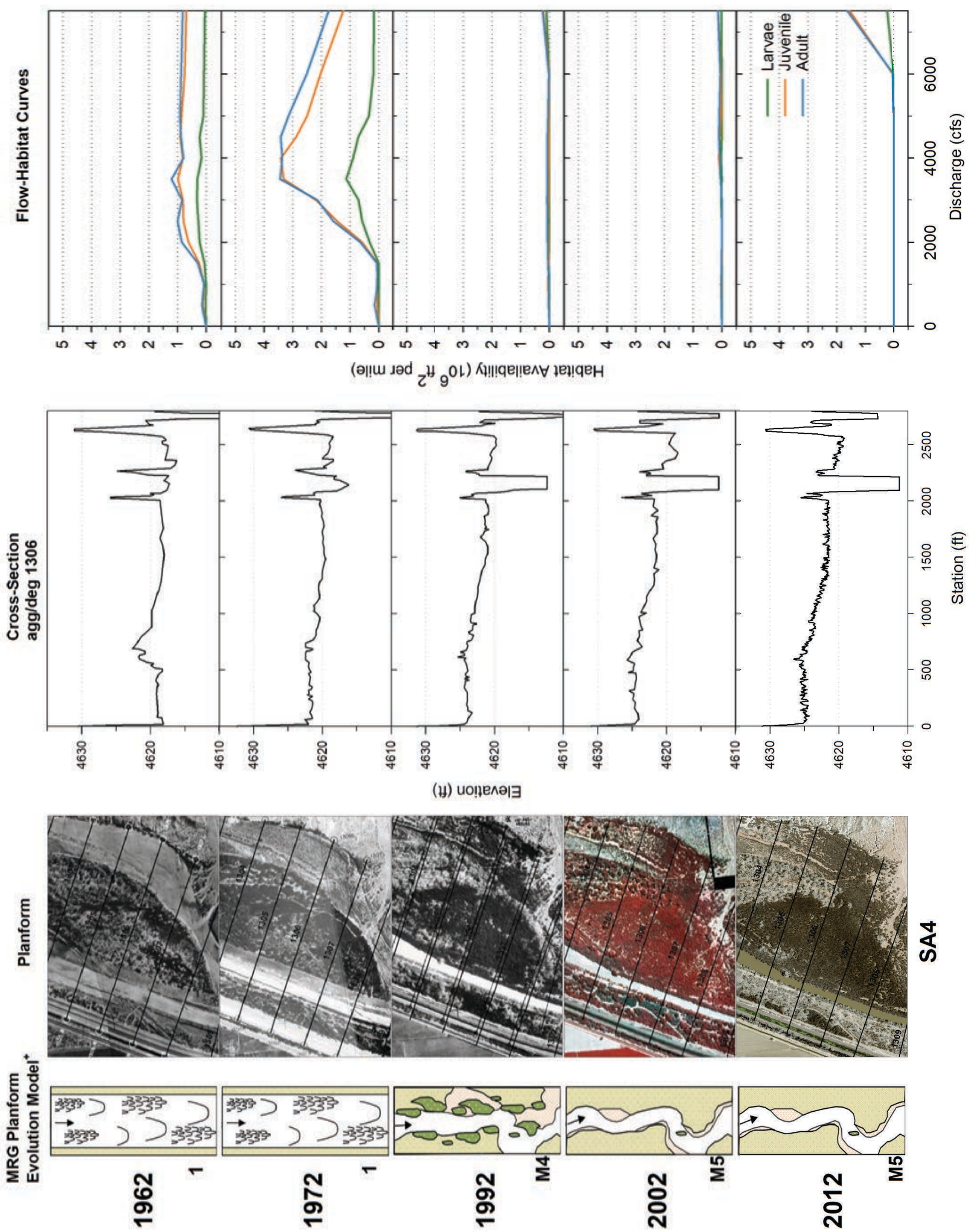


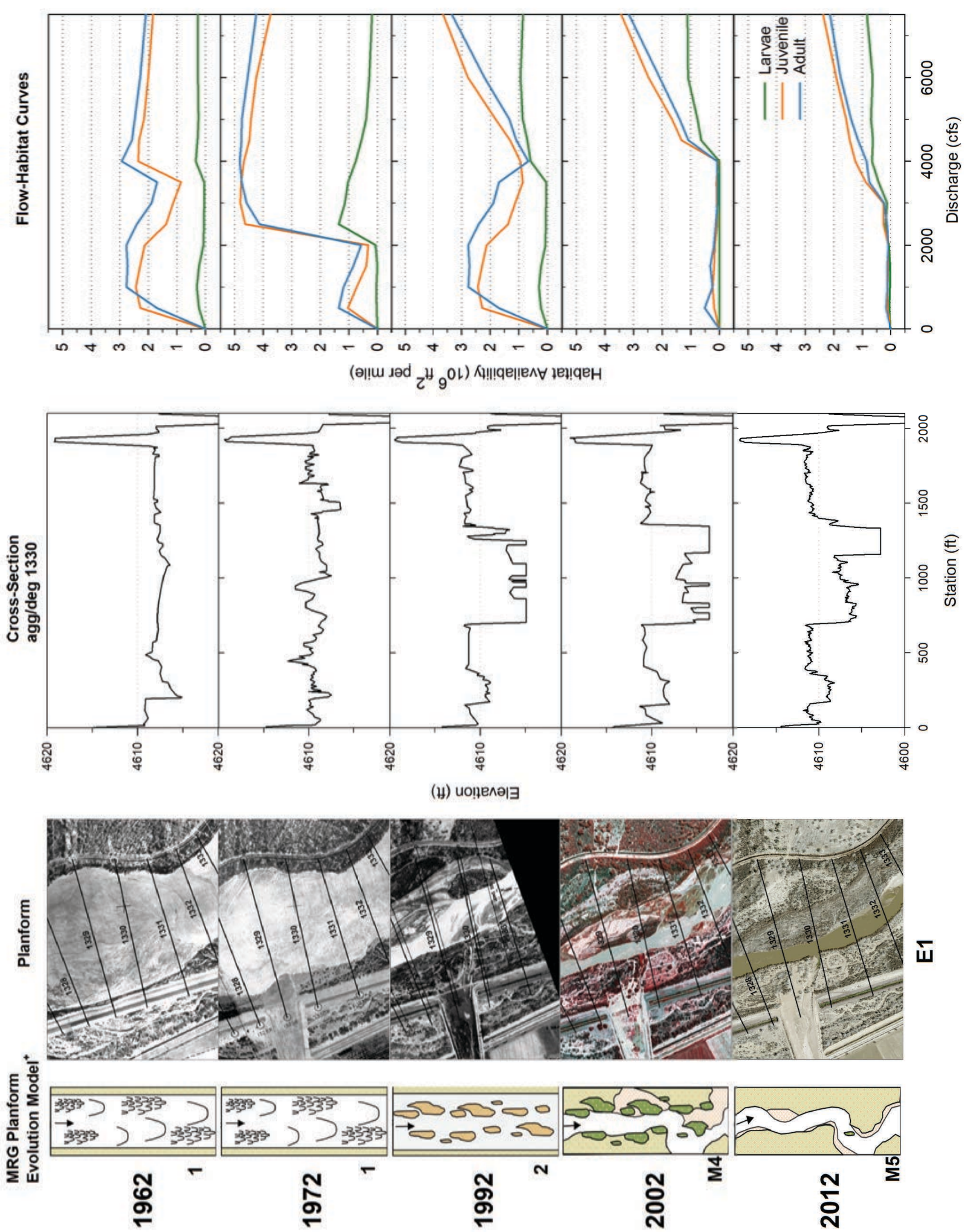

Figure A-1. Flow-habitat curves for the San Acacia Reach (combined subreaches). Curves are shown through time top to bottom (1962–2012). Line colors represent the primary life-stages of the Rio Grande Silvery Minnow. Habitat availability was normalized by reach length

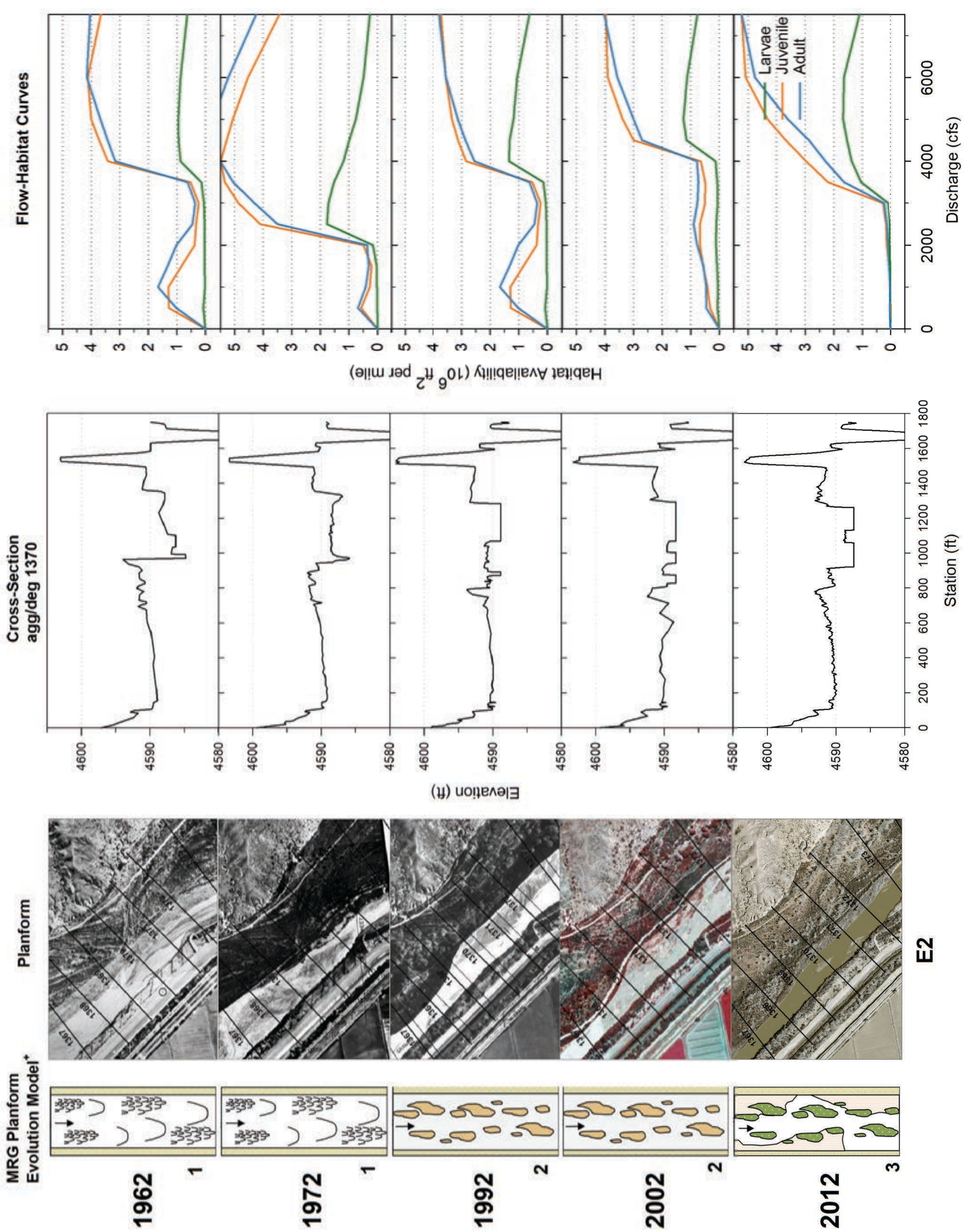

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1237
Flow-Habitat Curves
1962
1972
1992
2002
2012
1
1
1
3
M4
SA1
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

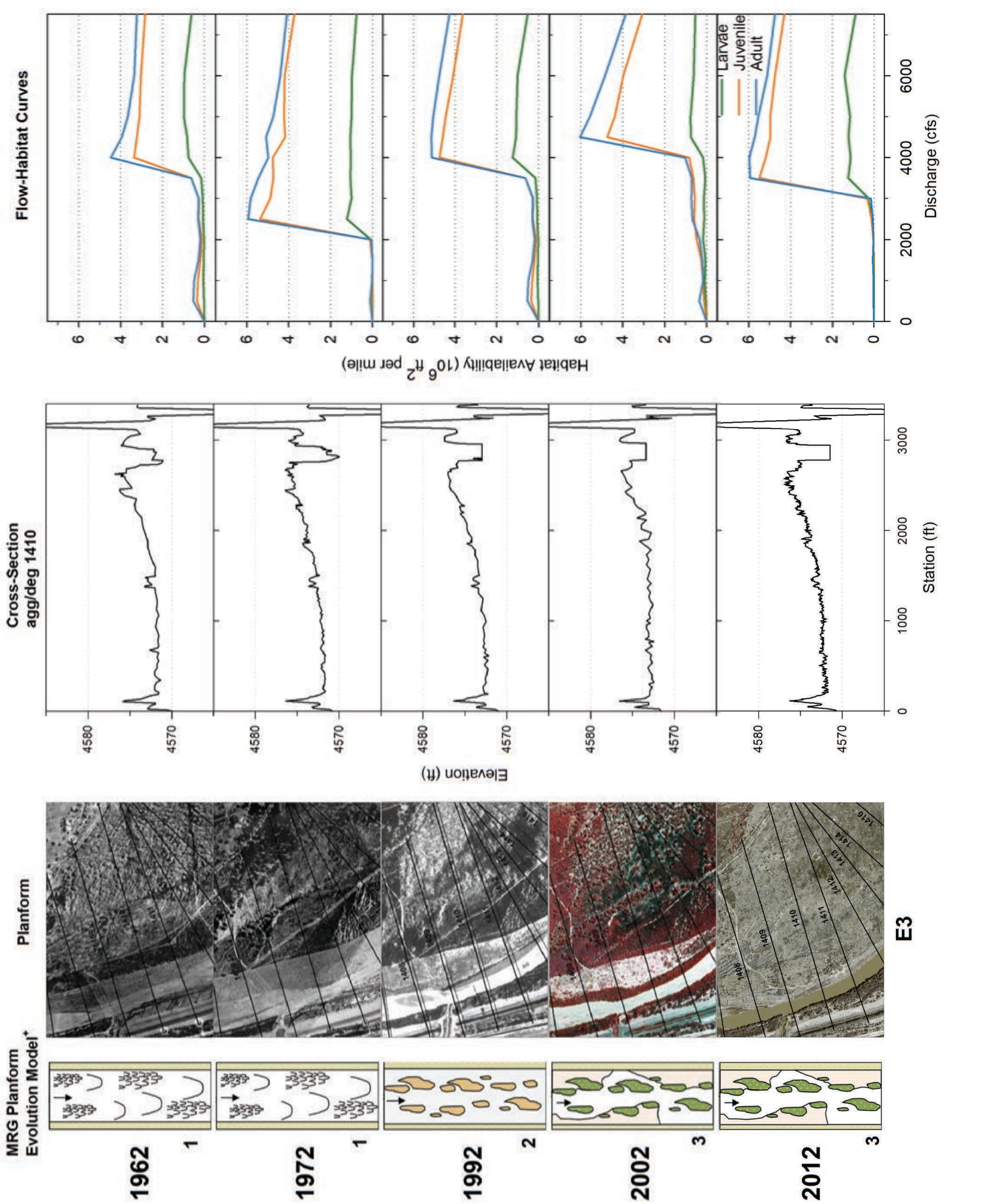

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1246
Flow-Habitat Curves
1962
1
1972
1
1992
3
2002
M4
2012
M5
SA2
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

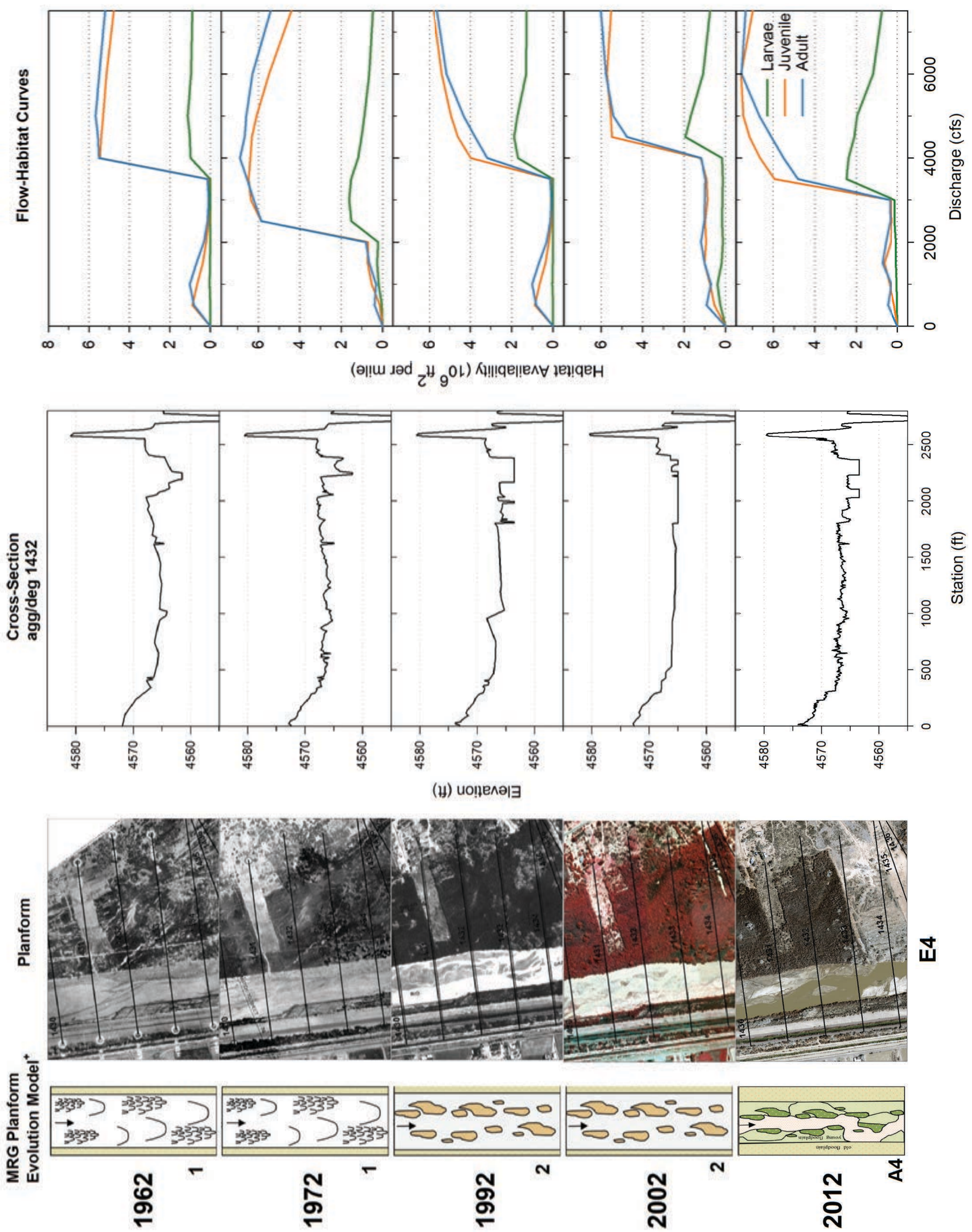

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1282
Flow-Habitat Curves
1962
1
1972
1
1992
1
2002
2
2012
3
SA3
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

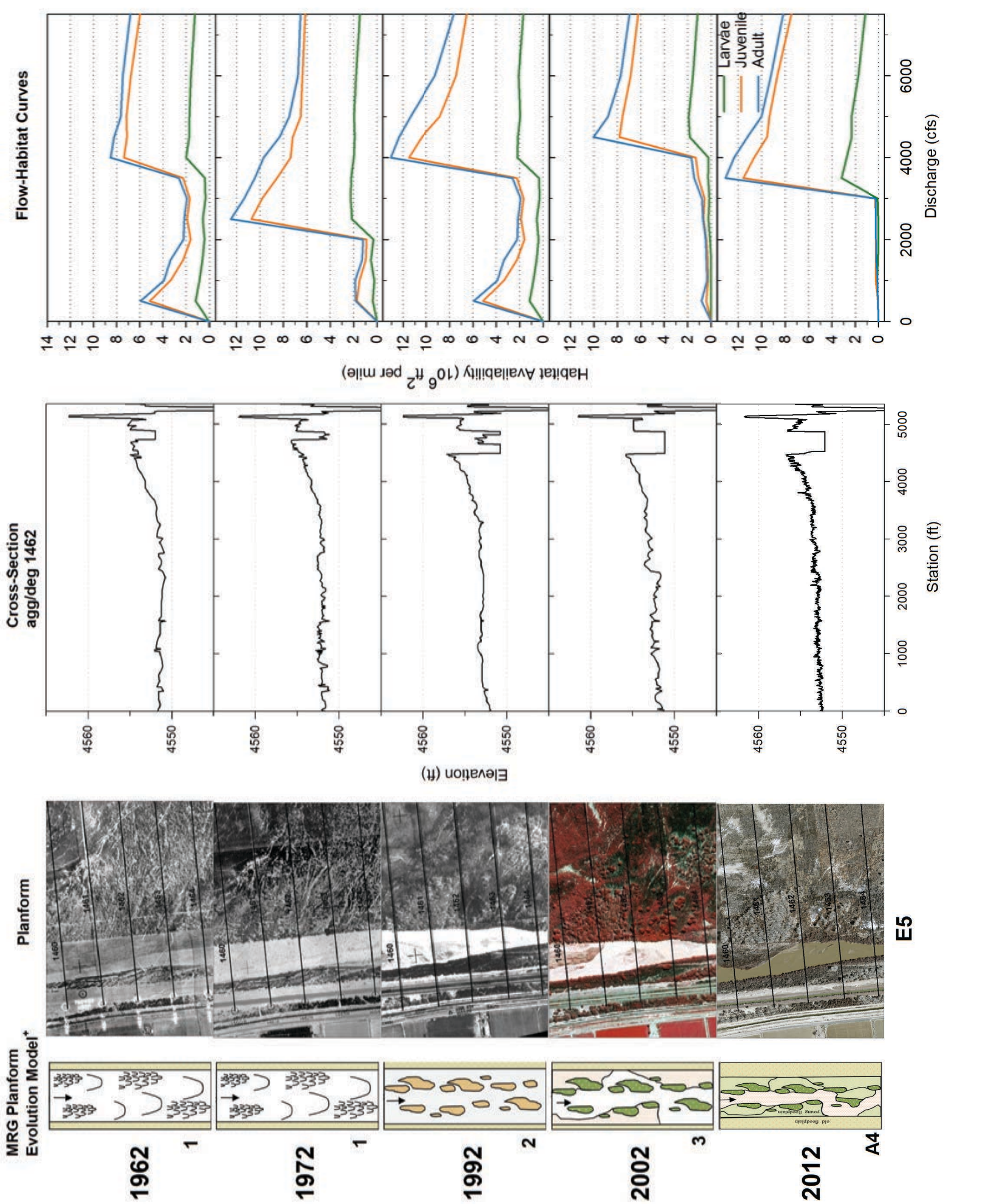

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1306
Flow-Habitat Curves
1962
1972
1992
2002
2012
1
1
M4
M5
M5
SA4
Elevation (ft)
Station (ft)
Habitat Availability (10$^6$ ft$^2$ per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

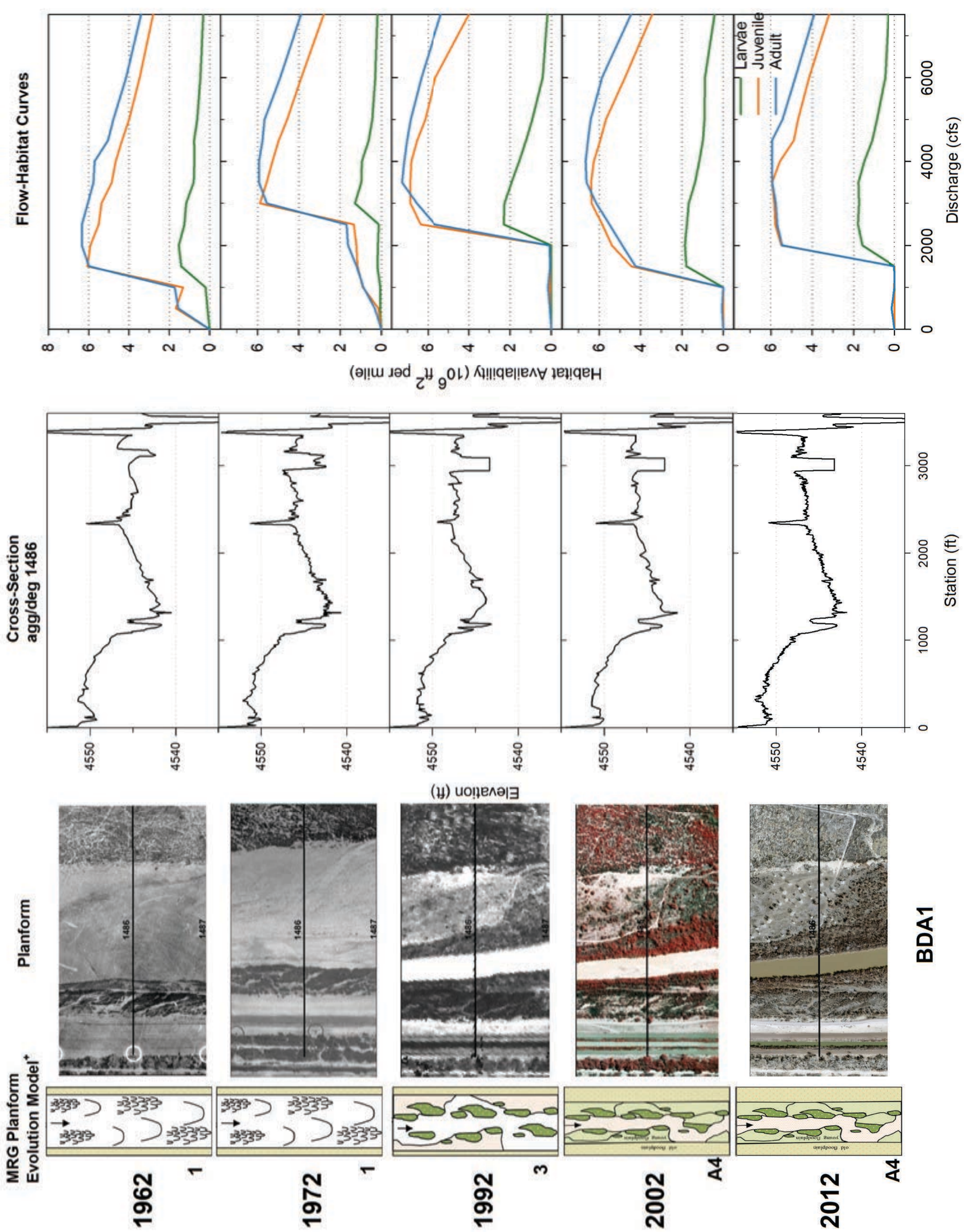

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1330
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
M4
2012
M5
E1
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

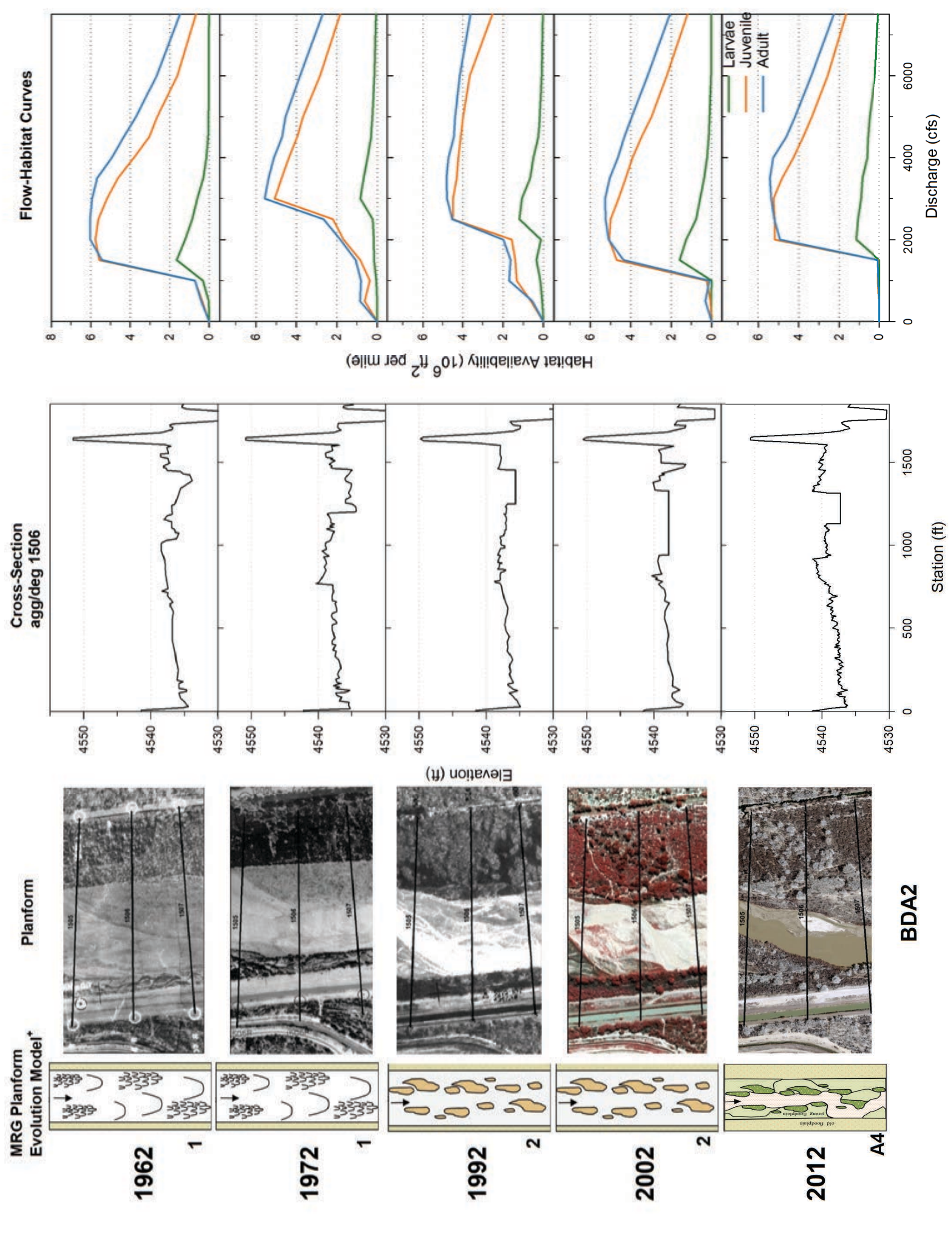

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1370
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
2
2012
3
E2
Elevation (ft)
Station (ft)
Habitat Availability (10$^6$ ft$^2$ per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

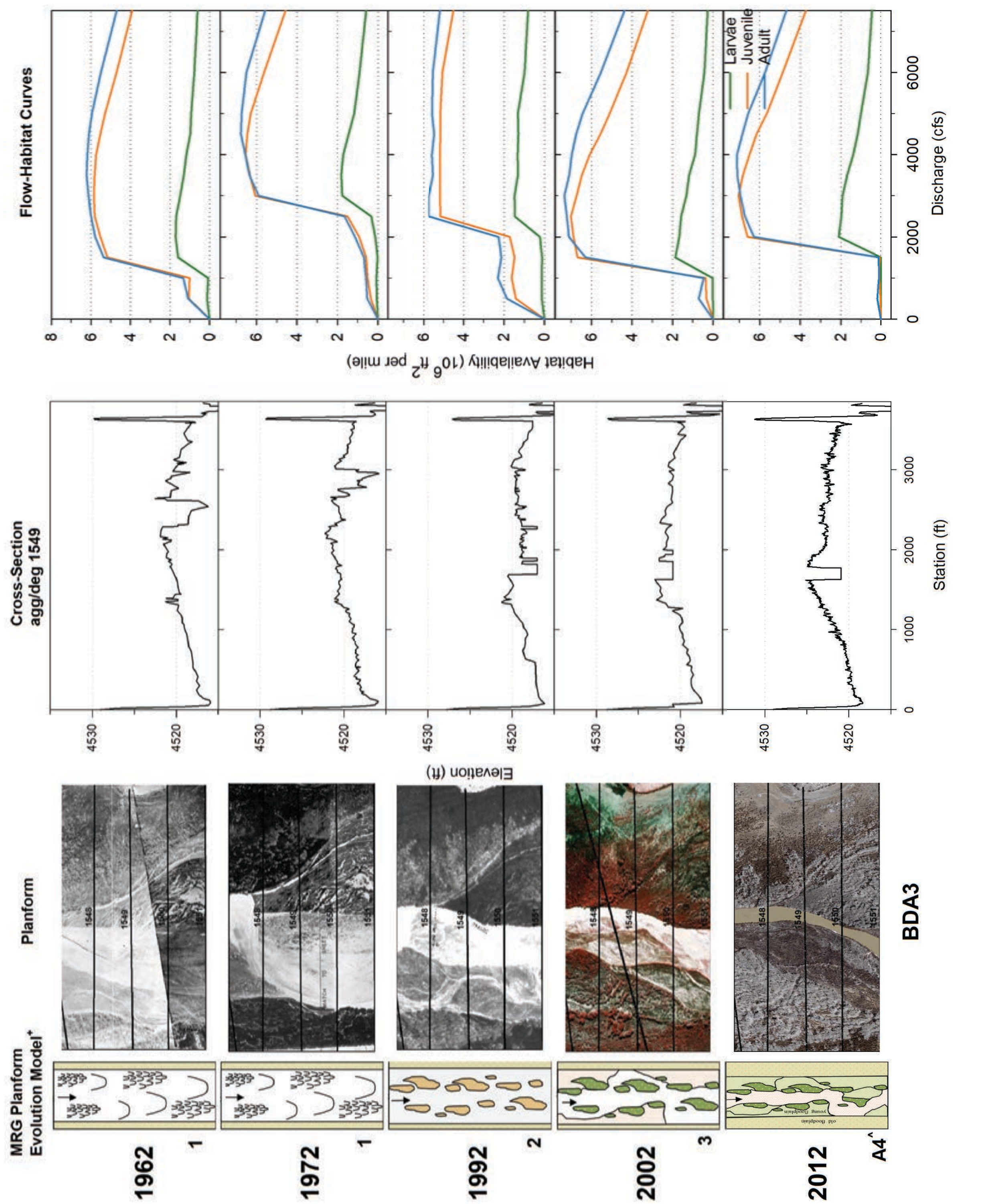

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1410
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
3
2012
3
E3
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

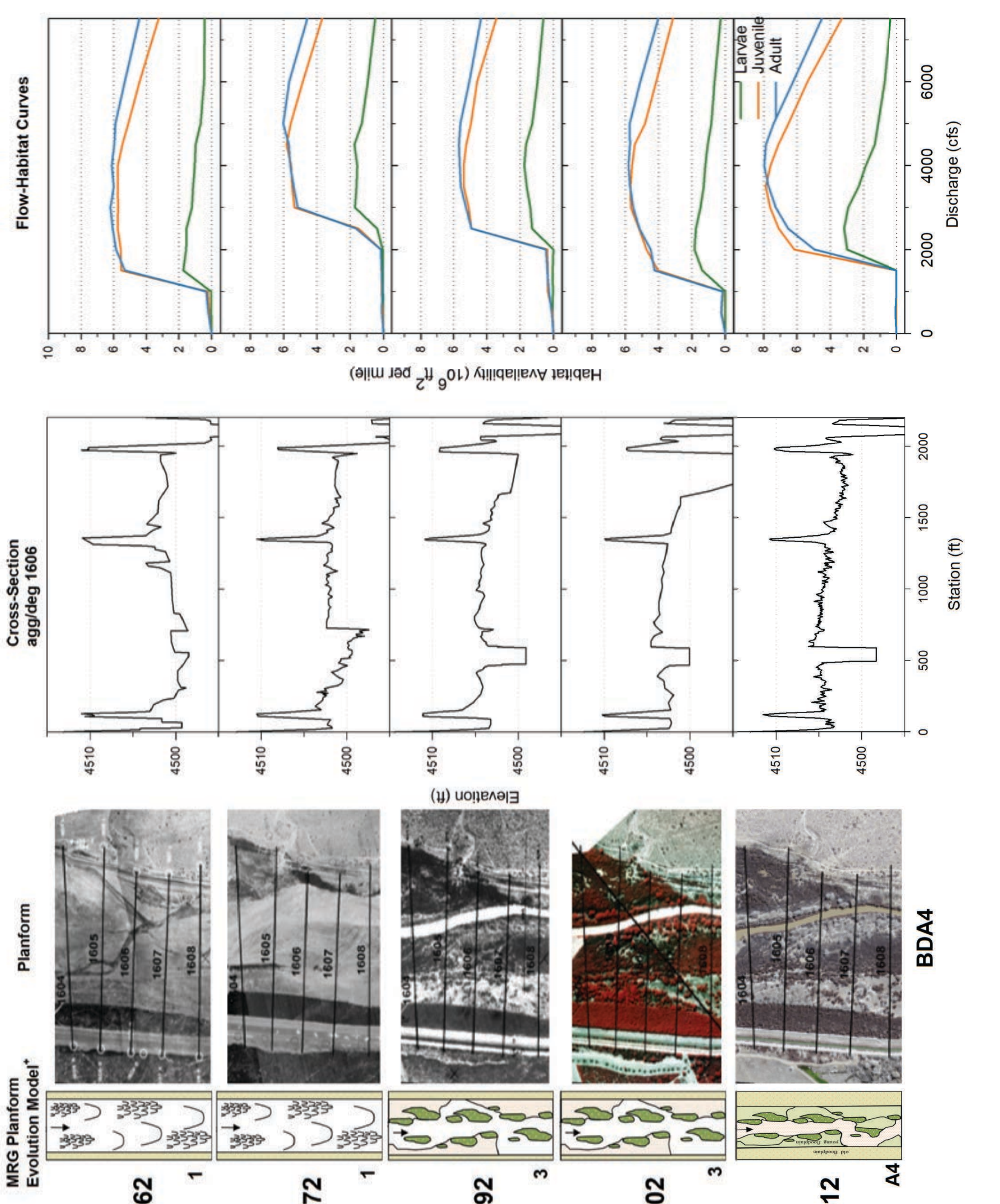

1962
1972
1992
2002
2012
1
1
2
2
A4
MRG Planform Evolution Model+
Planform
E4
Cross-Section agg/deg 1432
Elevation (ft)
Station (ft)
Flow-Habitat Curves
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

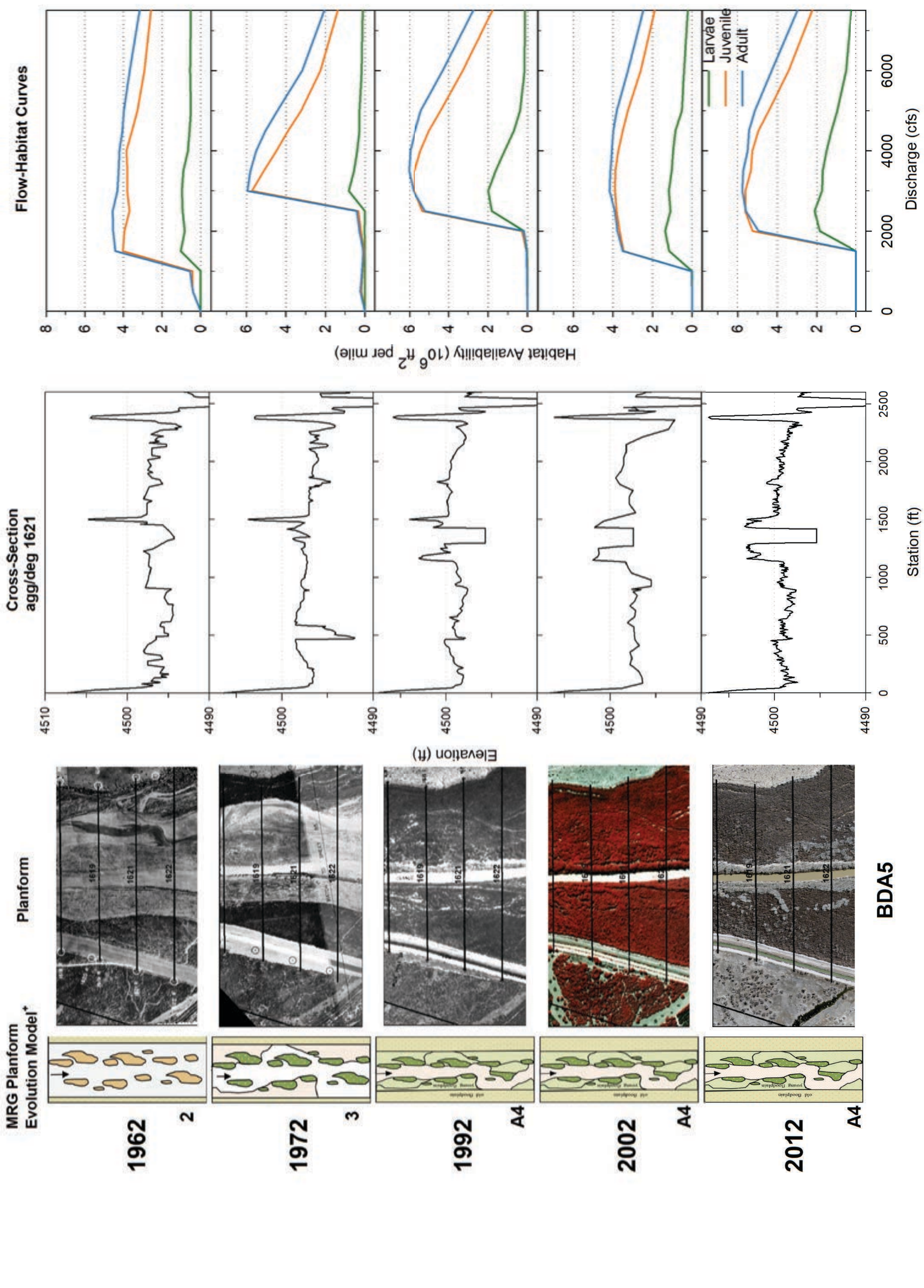

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1462
Flow-Habitat Curves
1962
1972
1992
2002
2012
1
1
2
3
A4
E5
Elevation (ft)
Station (ft)
Habitat Availability ($10^6$ ft$^2$ per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

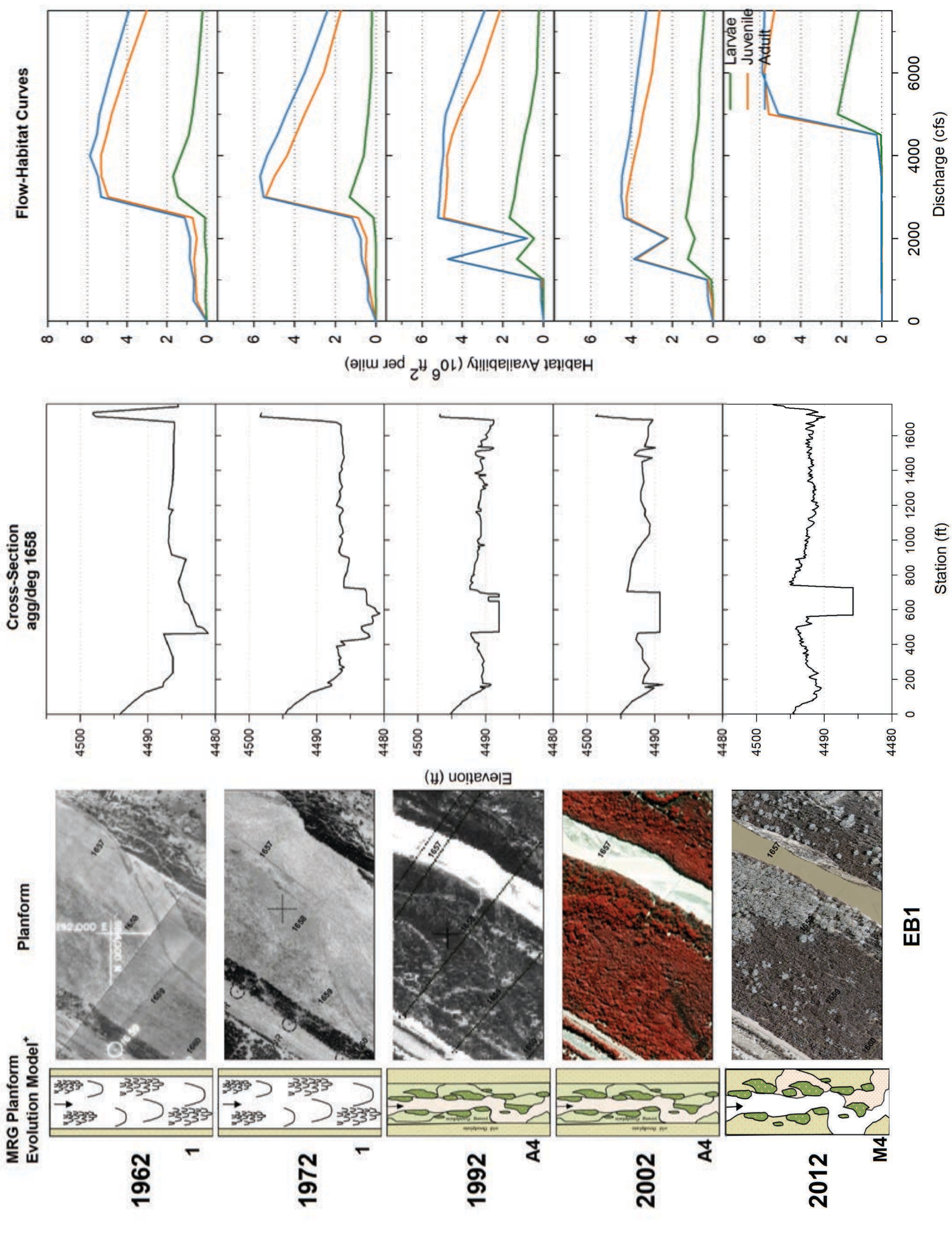

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1486
Flow-Habitat Curves
1962
1
1972
1
1992
3
2002
A4
2012
A4
BDA1
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

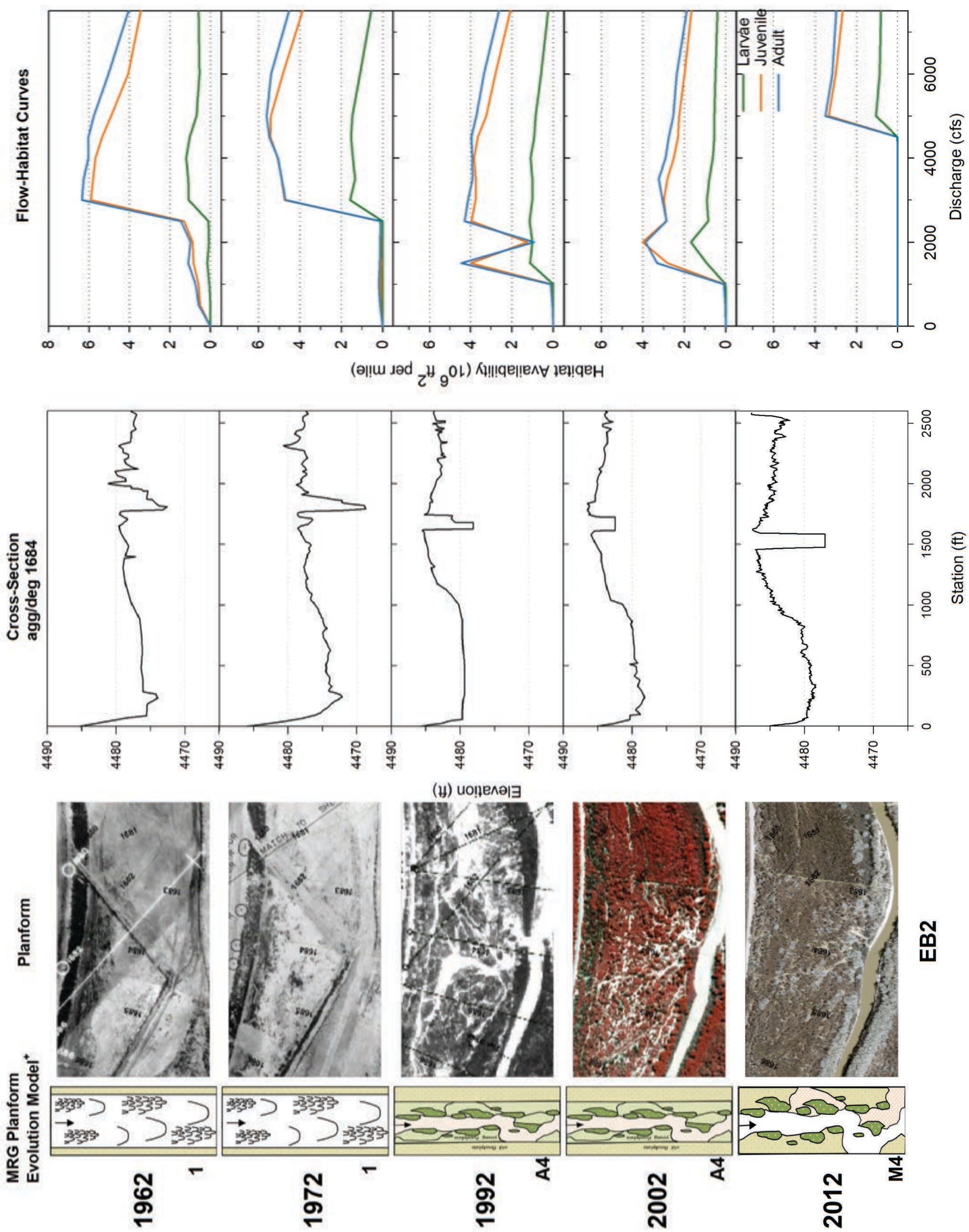

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1506
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
2
2012
A4
BDA2
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


MRG Planform Evolution Model+
Planform
Cross-Section agg/deg 1549
Flow-Habitat Curves
1962
1
1972
1
1992
2
2002
3
2012
A4^
BDA3
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

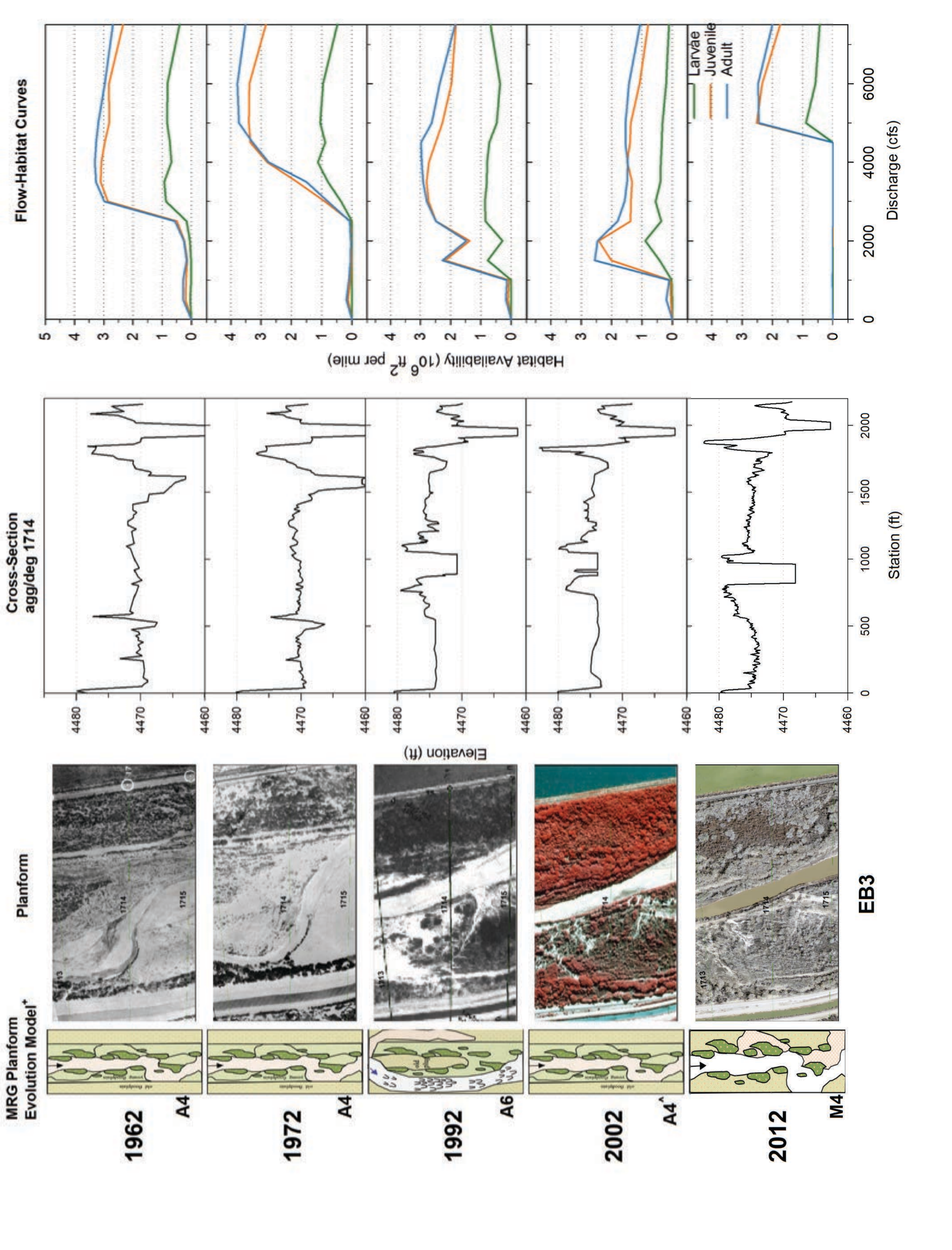

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1606
Flow-Habitat Curves
1962
1
1972
1
1992
3
2002
3
2012
A4
BDA4
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

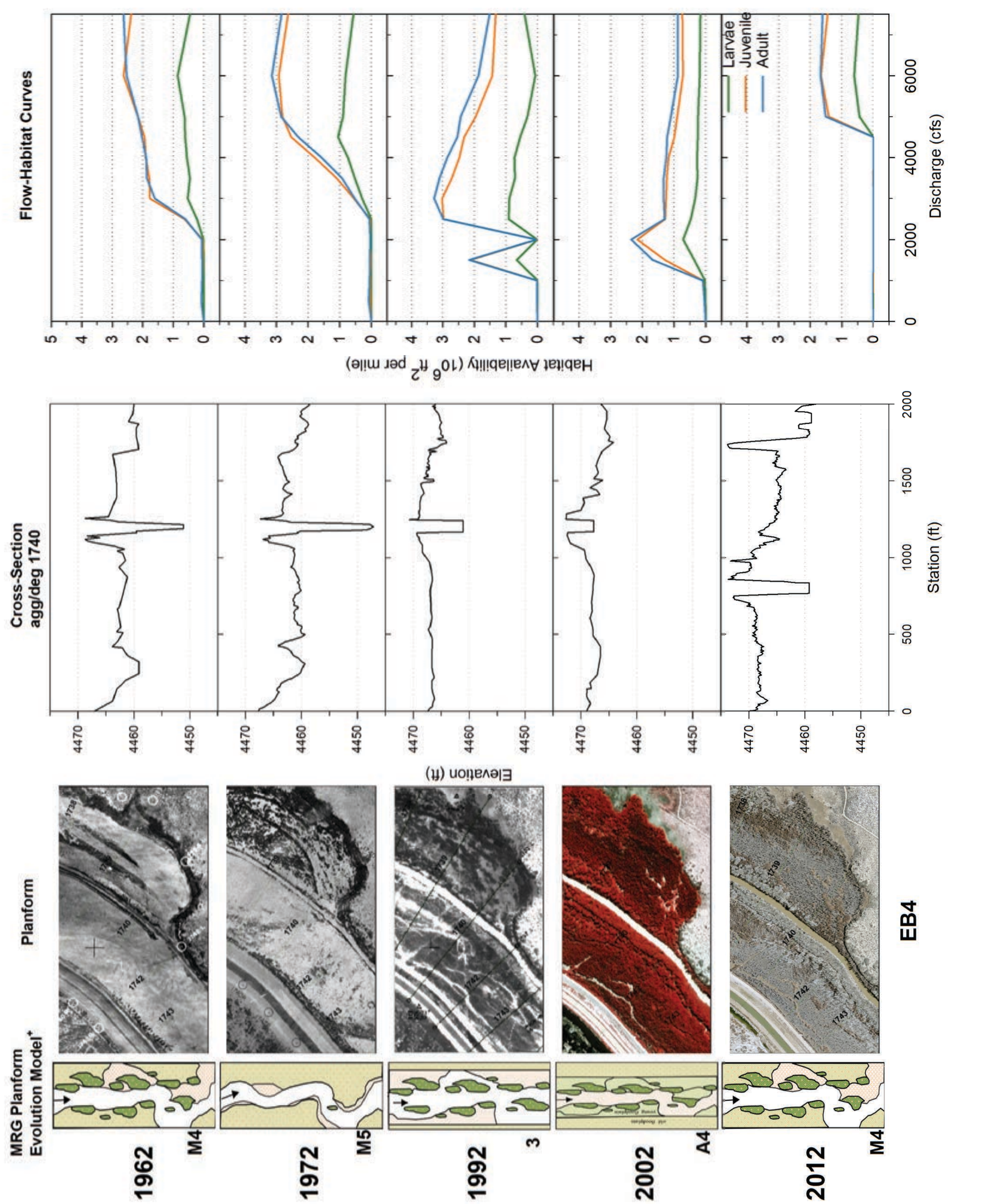

MRG Planform Evolution Model*
Planform
Cross-Section agg/deg 1621
Flow-Habitat Curves
1962
2
1972
3
1992
A4
2002
A4
2012
A4
BDA5
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


Flow-Habitat Curves
Cross-Section agg/deg 1658
Planform
MRG Planform Evolution Model*
1962
1972
1992
2002
2012
1
1
A4
A4
M4
EB1
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Elevation (ft)
Station (ft)
Larvae
Juvenile
Adult



1962
1972
1992
2002
2012
1
1
A4
A4
M4
MRG Planform Evolution Model*
Planform
EB2
Cross-Section agg/deg 1684
Elevation (ft)
Station (ft)
Flow-Habitat Curves
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


MRG Planform Evolution Model+
Planform
Cross-Section agg/deg 1714
Flow-Habitat Curves
1962
A4
1972
A4
1992
A6
2002
A4^
2012
M4
EB3
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult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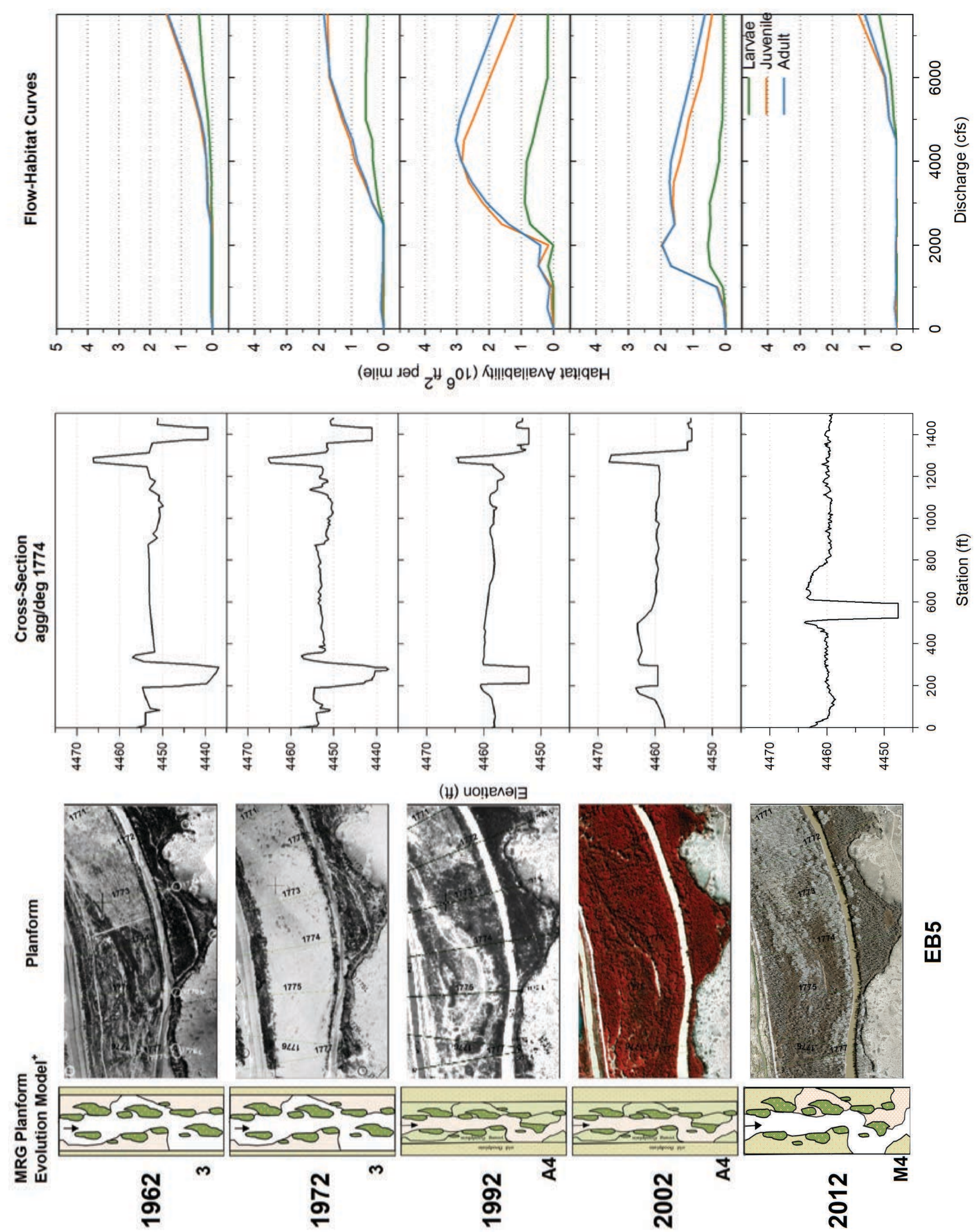
Planform
Cross-Section agg/deg 1740
Flow-Habitat Curves
1962
M4
1972
M5
1992
3
2002
A4
2012
M4
EB4
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult


MRG Planform Evolution Model+
Planform
Cross-Section agg/deg 1774
Flow-Habitat Curves
1962
3
1972
3
1992
A4
2002
A4
2012
M4
EB5
Elevation (ft)
Station (ft)
Habitat Availability (10^6 ft^2 per mile)
Discharge (cfs)
Larvae
Juvenile
Adult

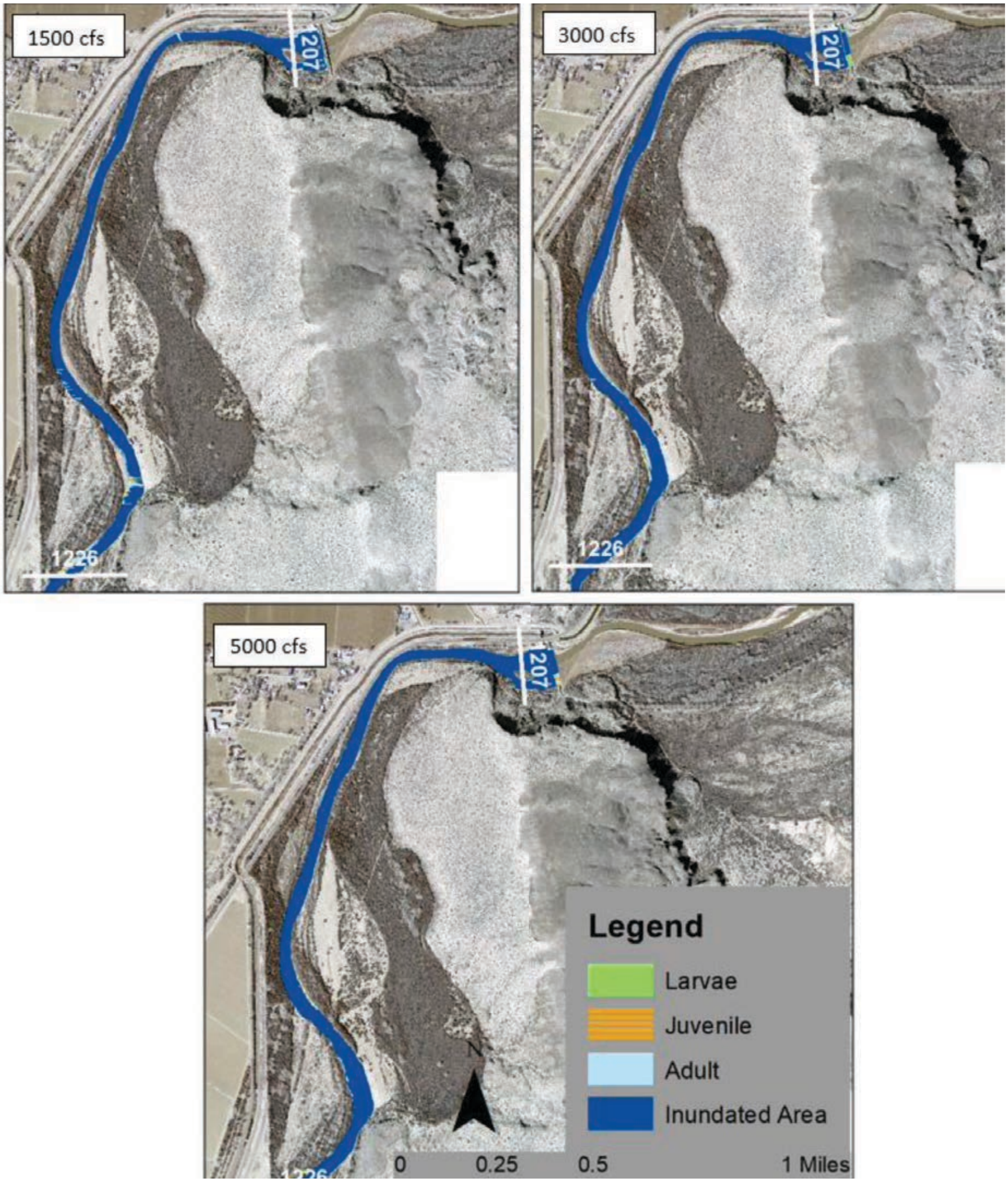



Figure A-21. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach SA1a (agg/deg 1207–1245).

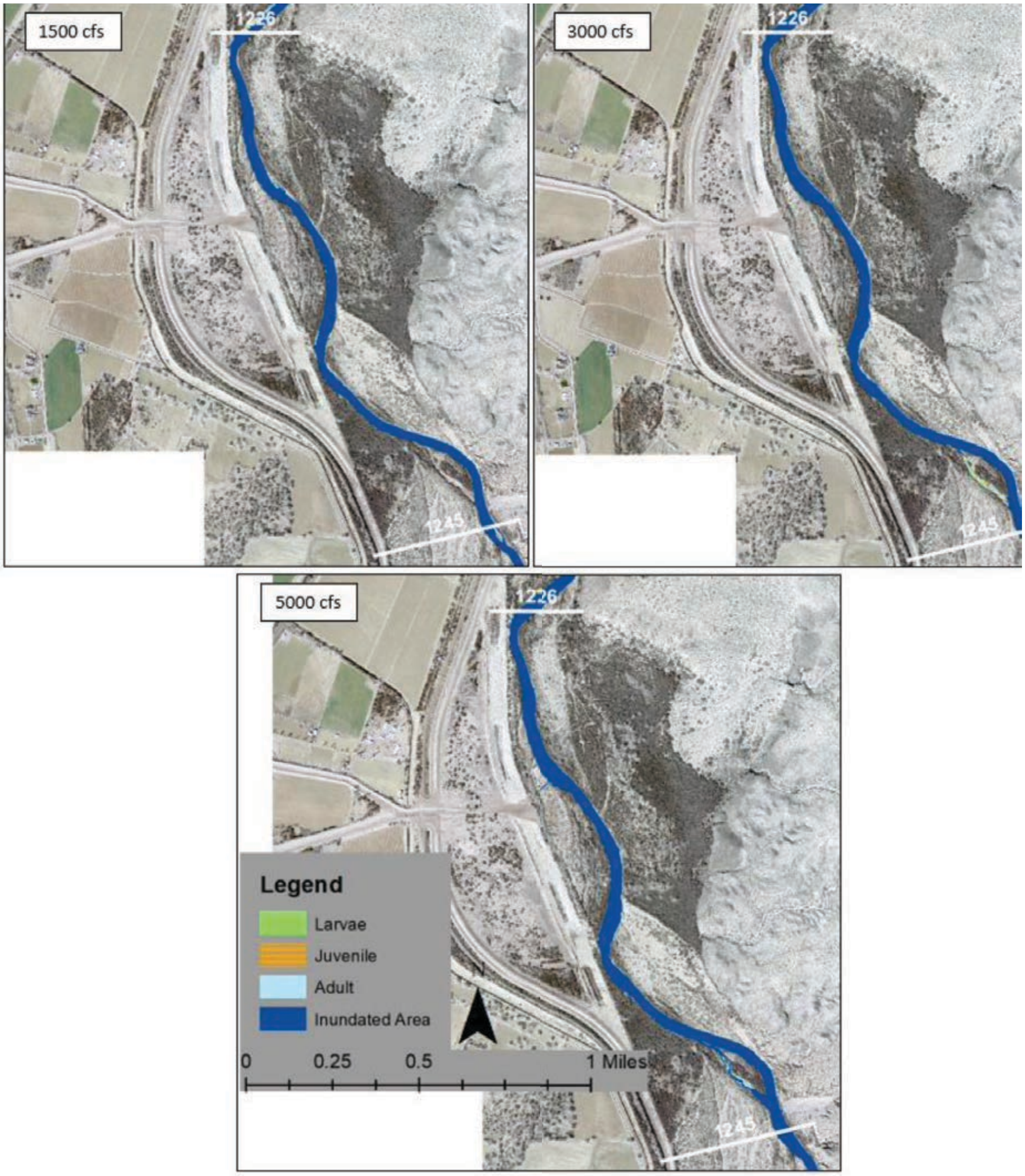
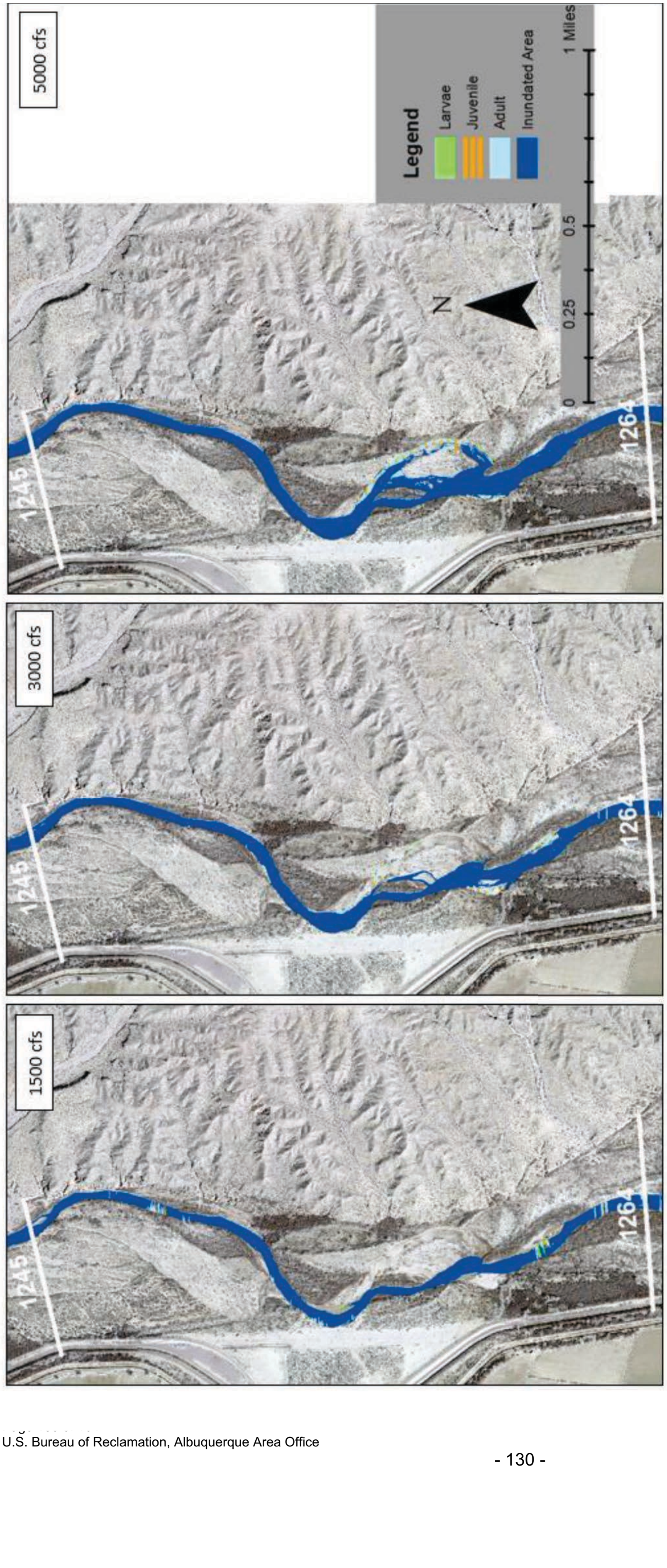
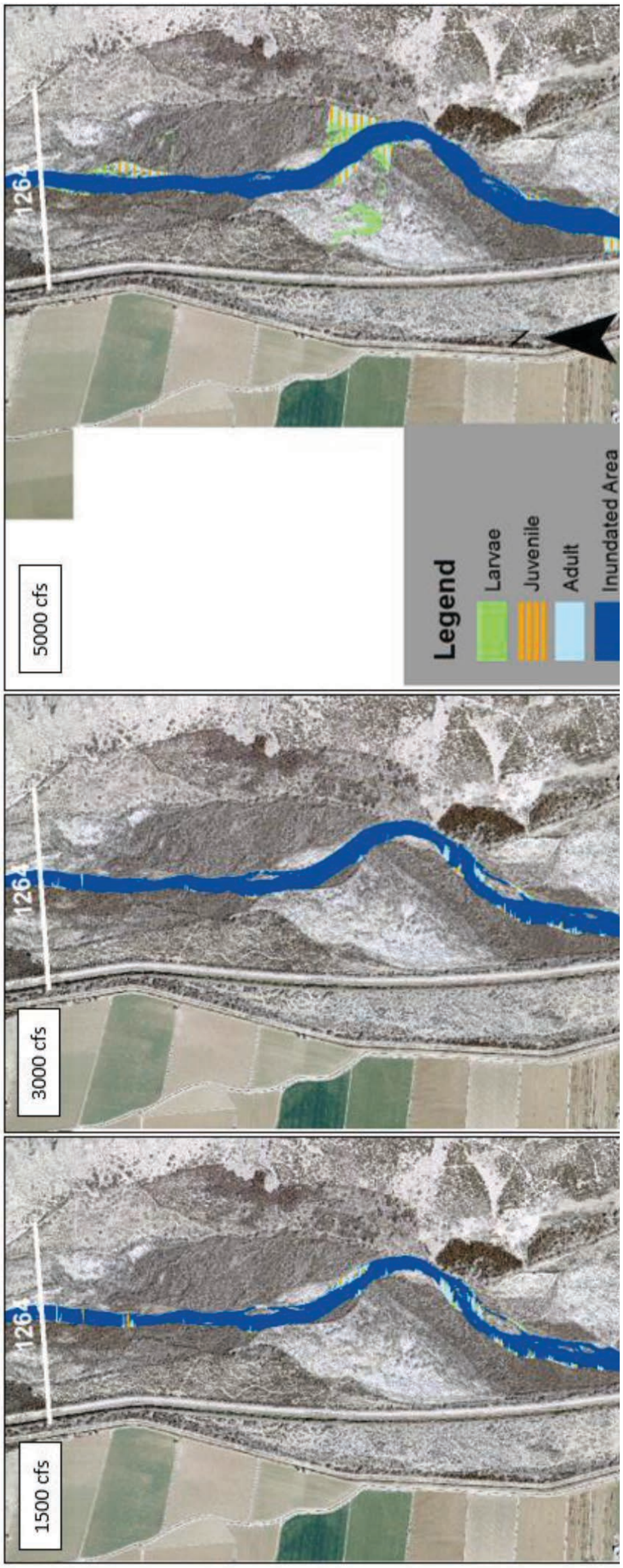
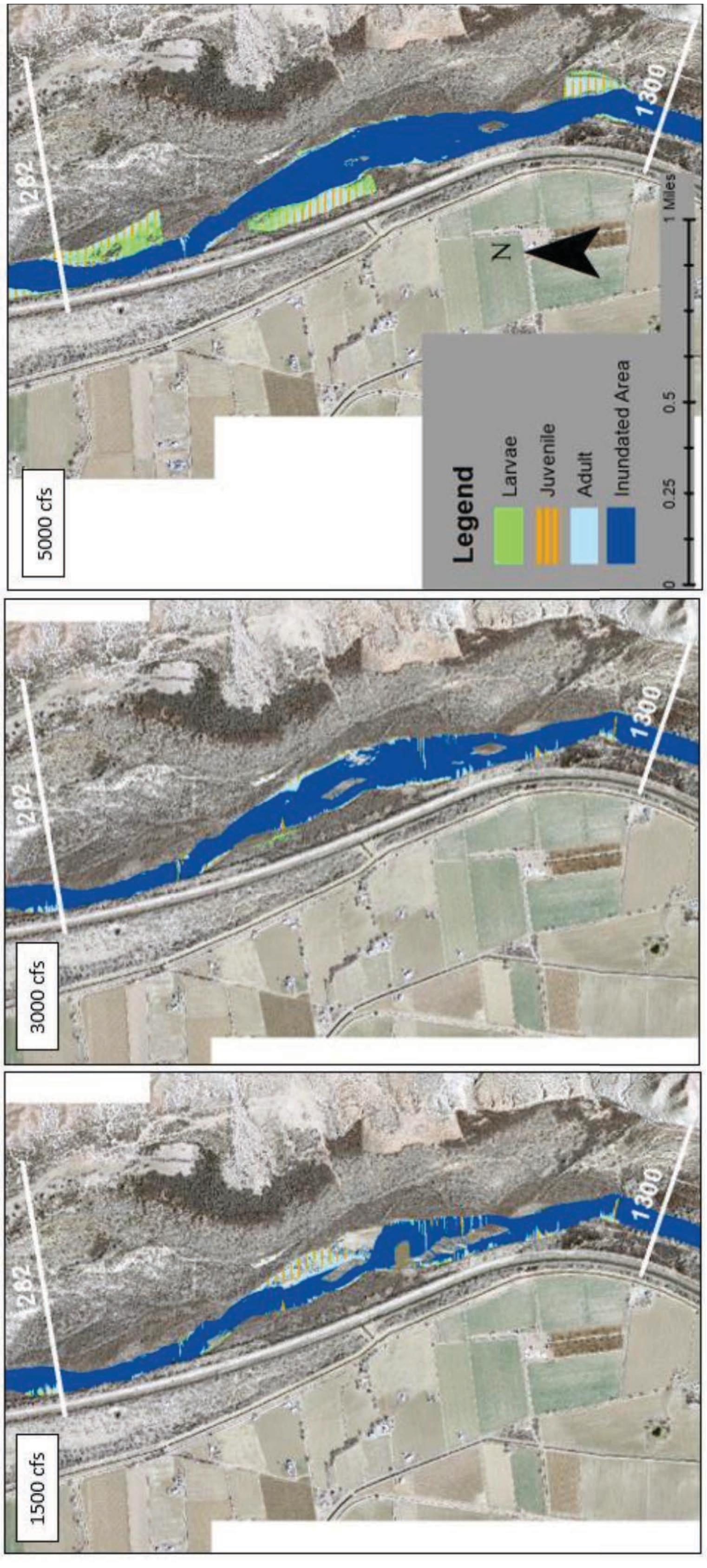



Figure A-22. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach SA1b (agg/deg 1207–1245).


5000 cfs
Legend
Larvae
Juvenile
Adult
Inundated Area
0
0.25
0.5
1 Miles
N
1245
1264
3000 cfs
1245
1264
1500 cfs
1245
1264


5000 cfs
3000 cfs
1500 cfs
1264
1264
1264
Legend
Larvae
Juvenile
Adult
Inundated Area


5000 cfs
3000 cfs
1500 cfs
282
1300
N
Legend
Larvae
Juvenile
Adult
Inundated Area
0
0.25
0.5
1 Miles

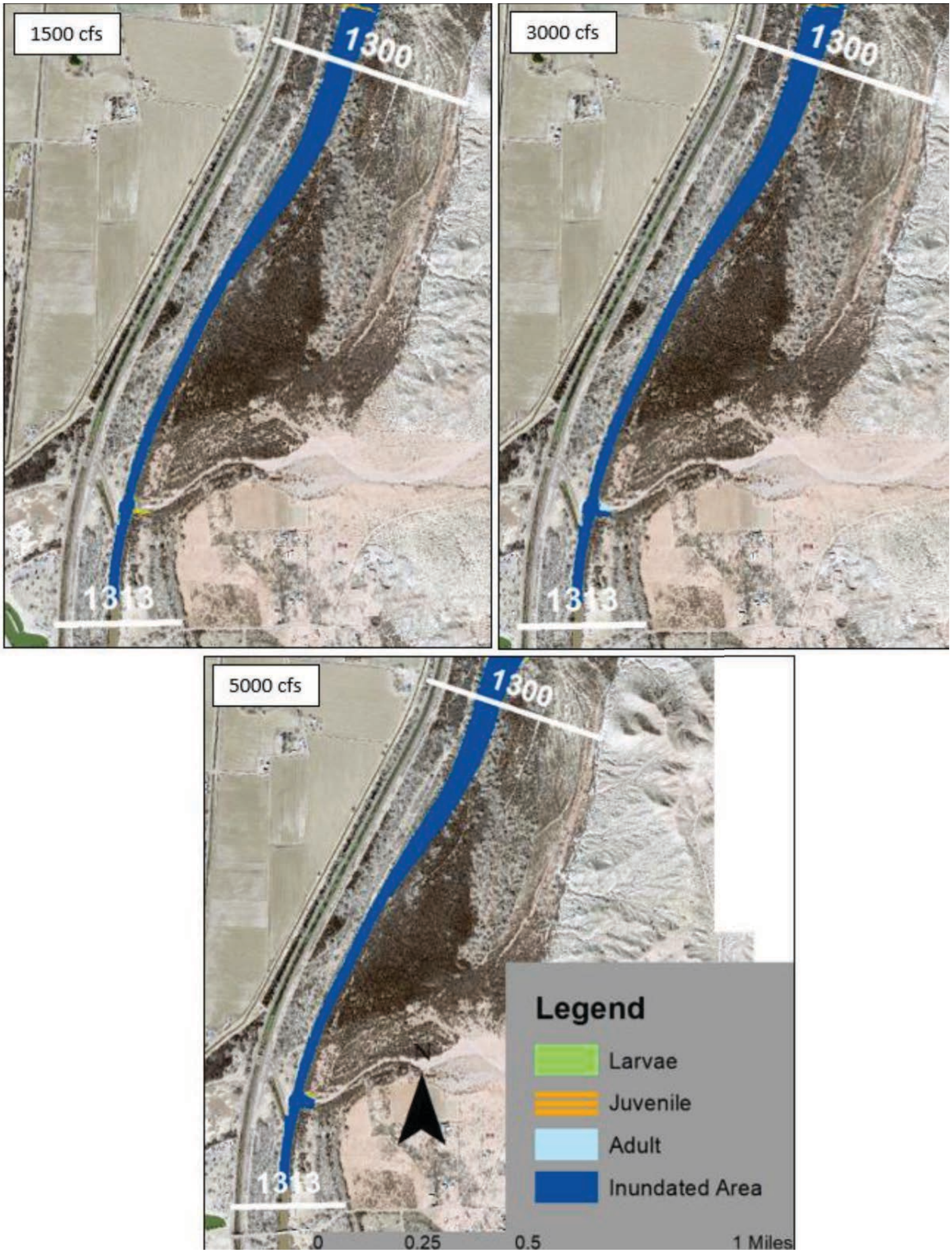



Figure A-26. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach SA4 (agg/deg 1300–1313).

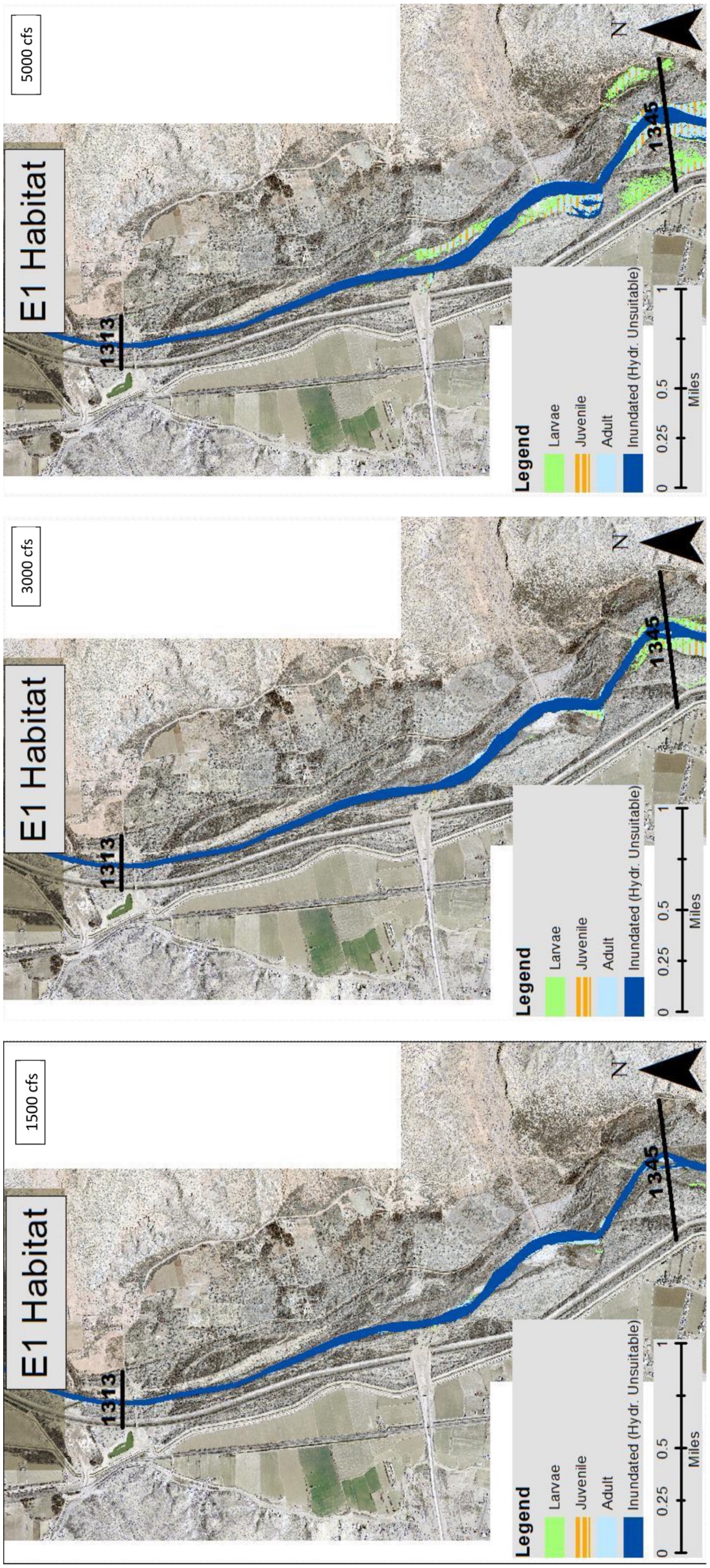



Figure A-27. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach E1 (agg/deg 1313–1345).

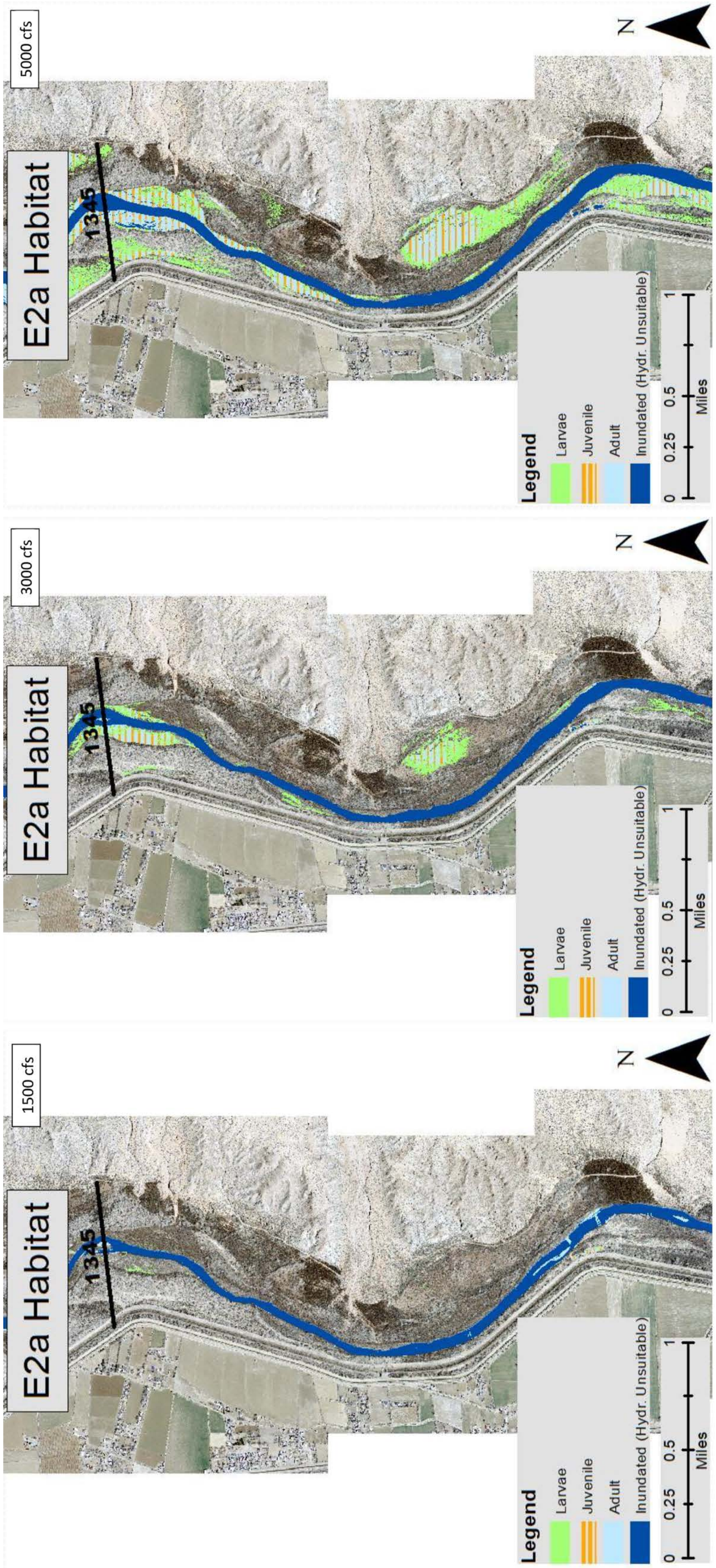



Figure A-28. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach E2a (agg/deg 1345–1397).

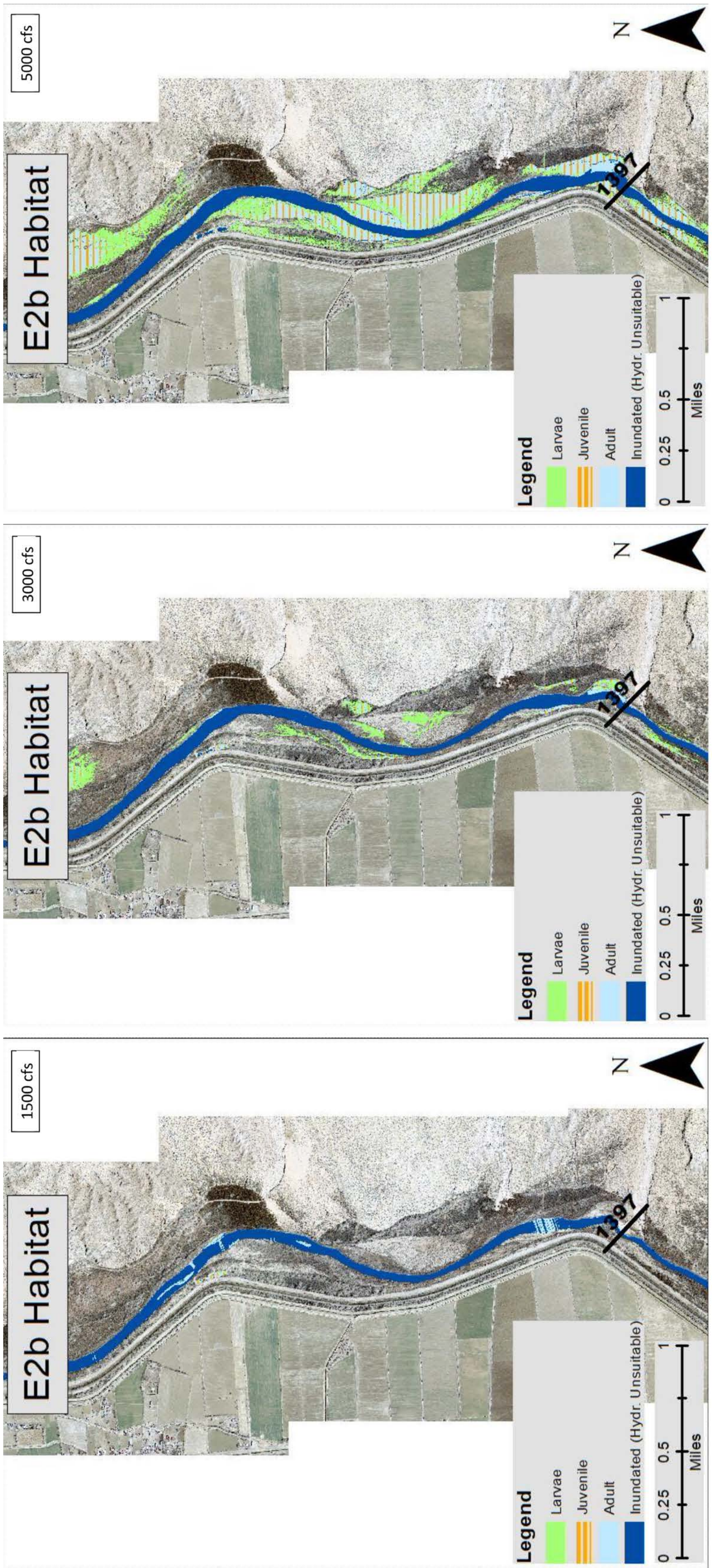



Figure A-29. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach E2b (agg/deg 1345–1397).

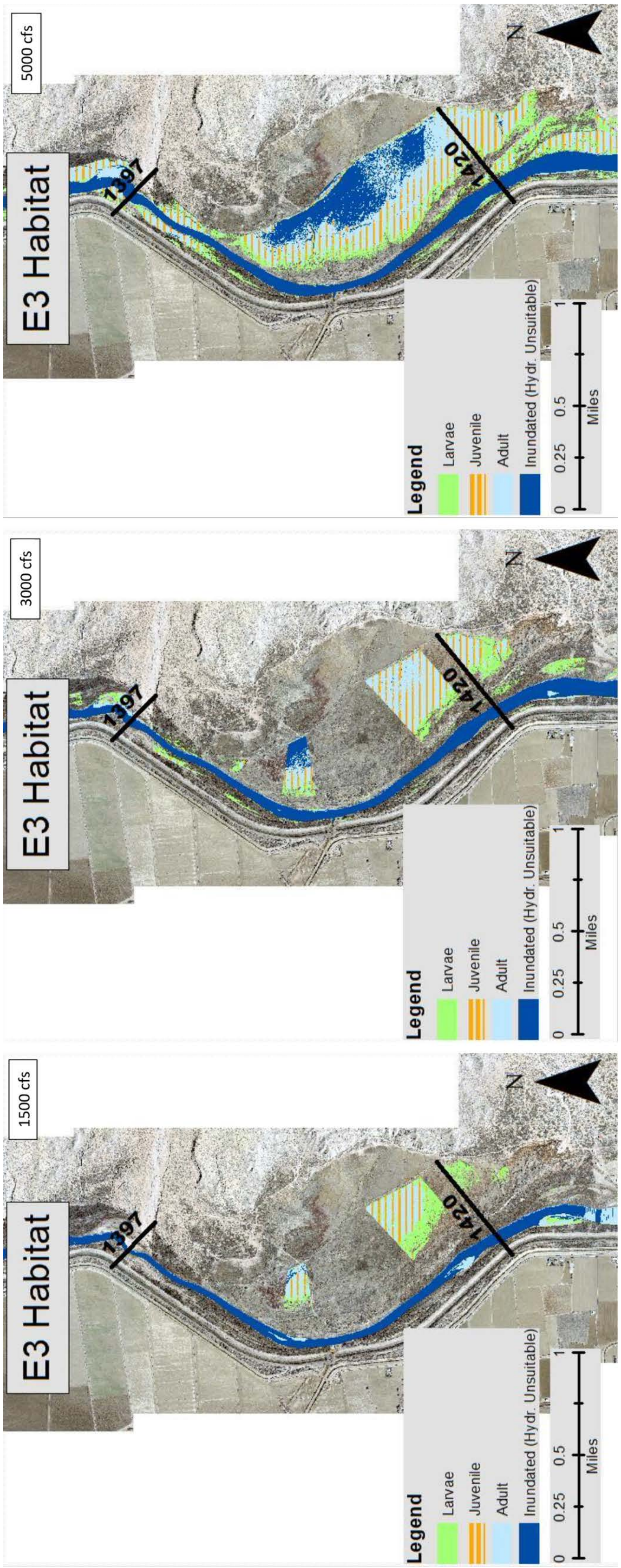



Figure A-30. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach E3 (agg/deg 1397–1420).

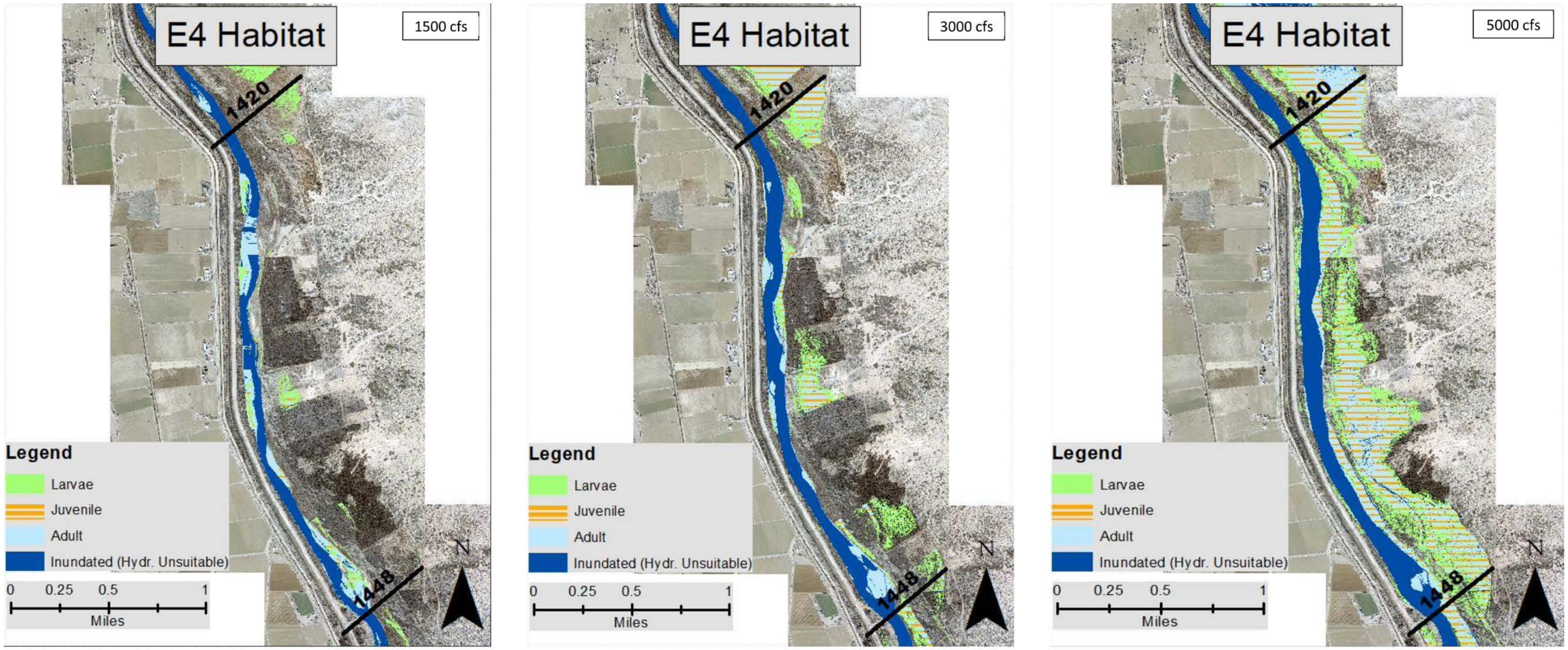



Figure A-31. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach E4 (agg/deg 1420–1448).

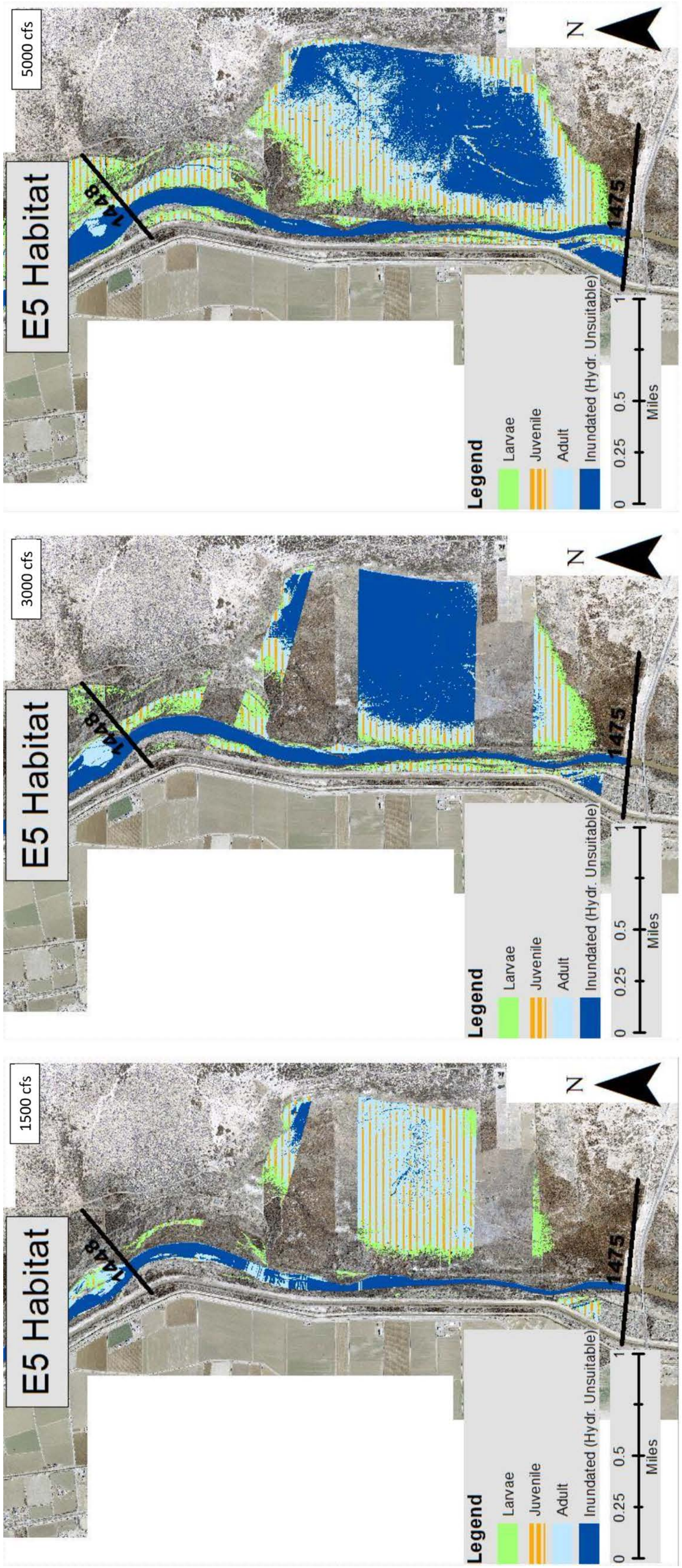



Figure A-32. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach E5 (agg/deg 1448–1475).

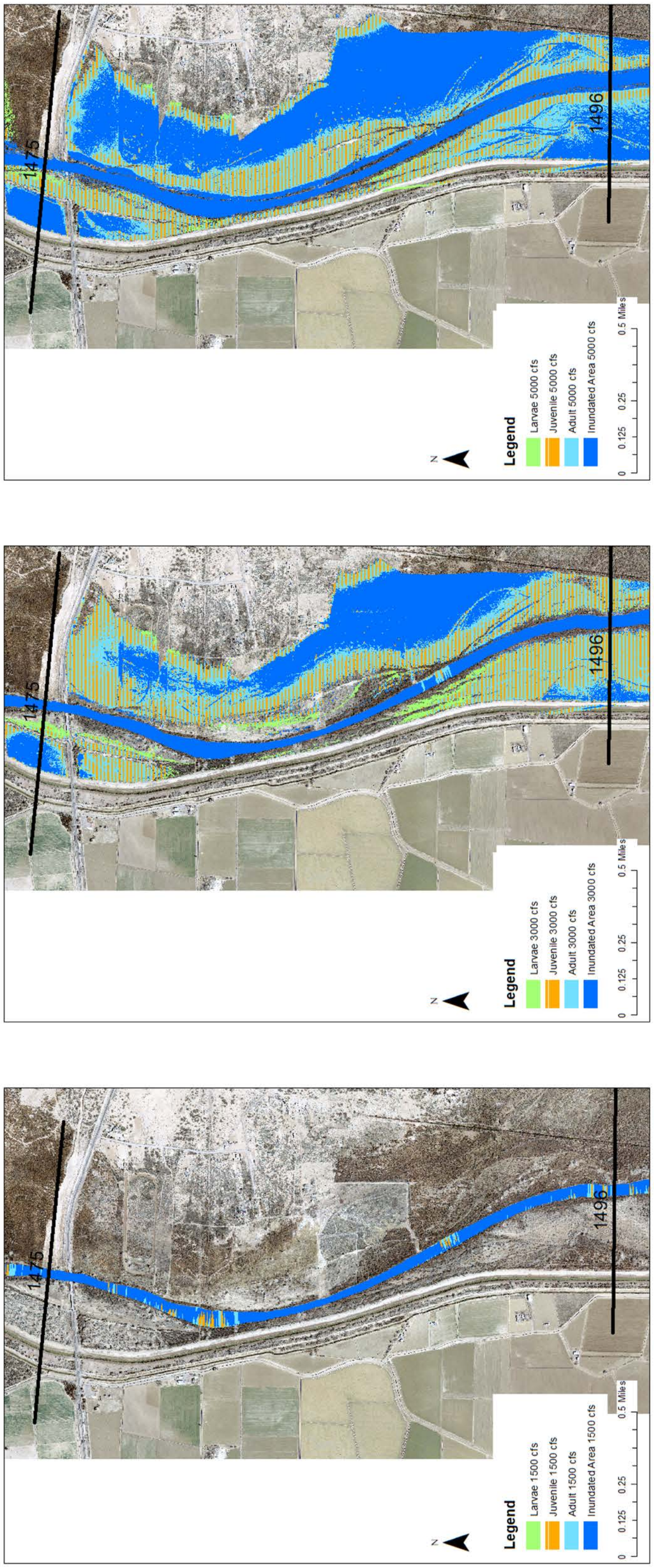
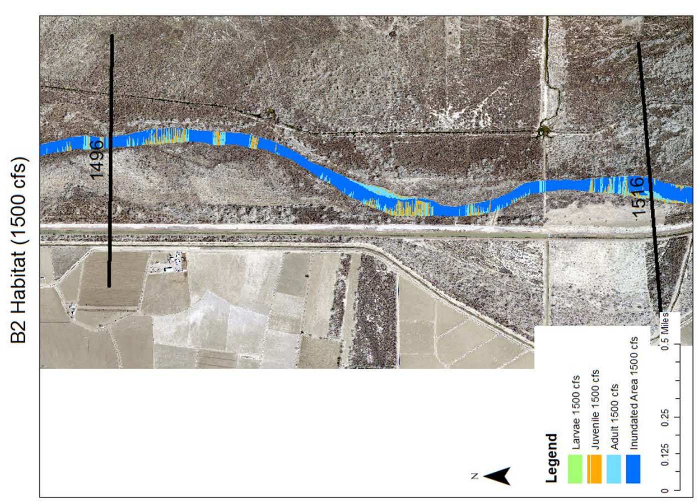
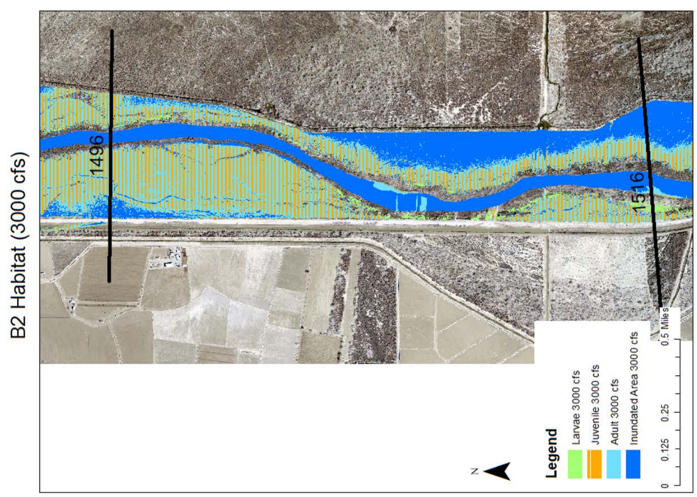



Figure A-33. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA1 (agg/deg 1475–1496).



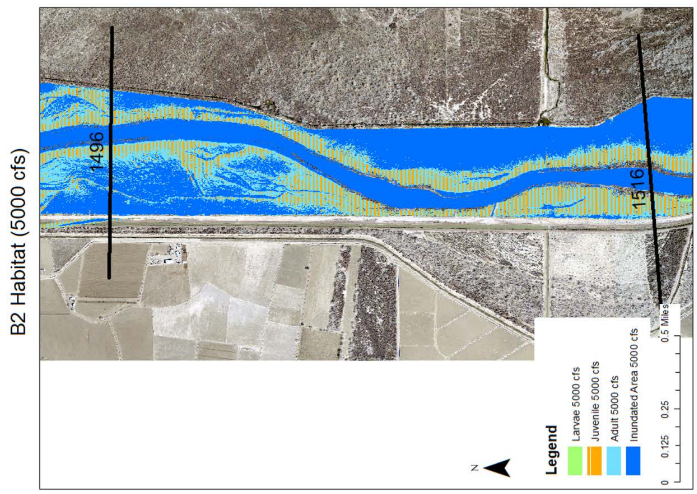




Figure A-34. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA2 (agg/deg 1496–1516).

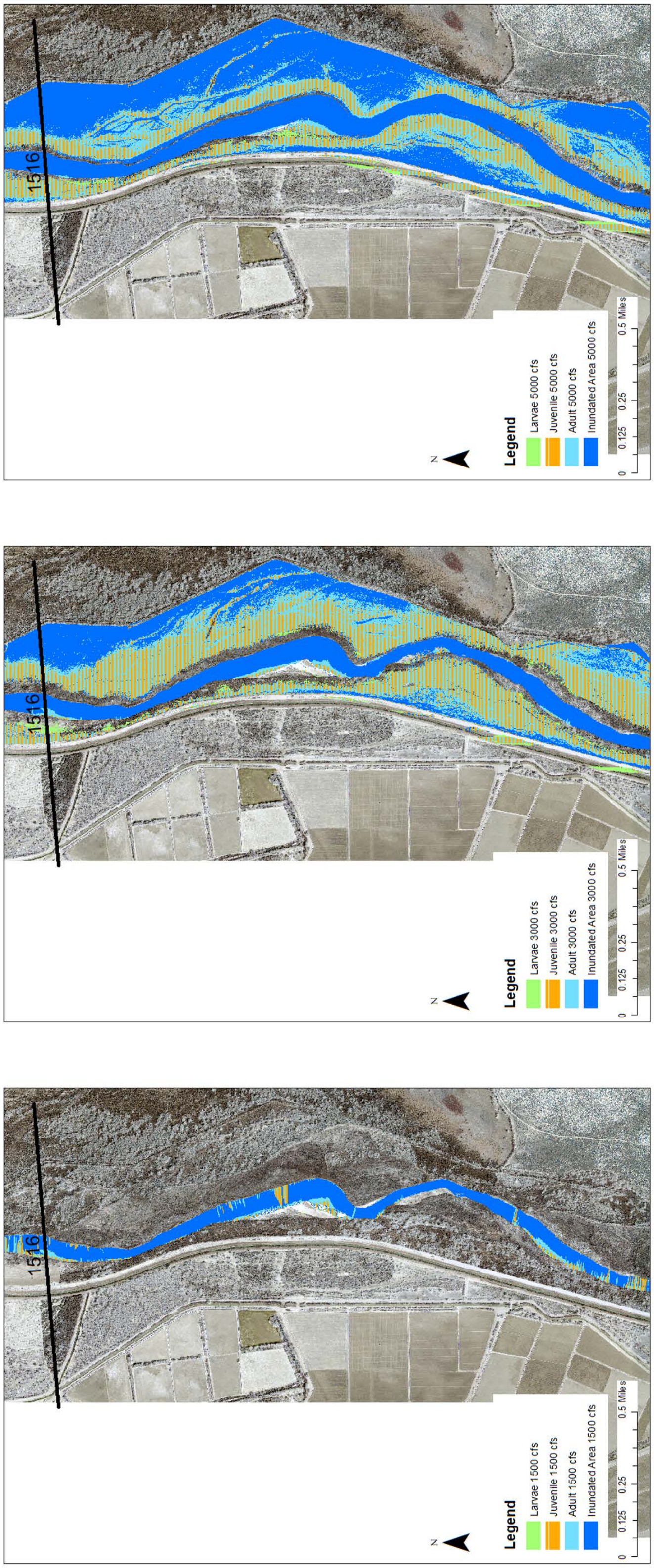
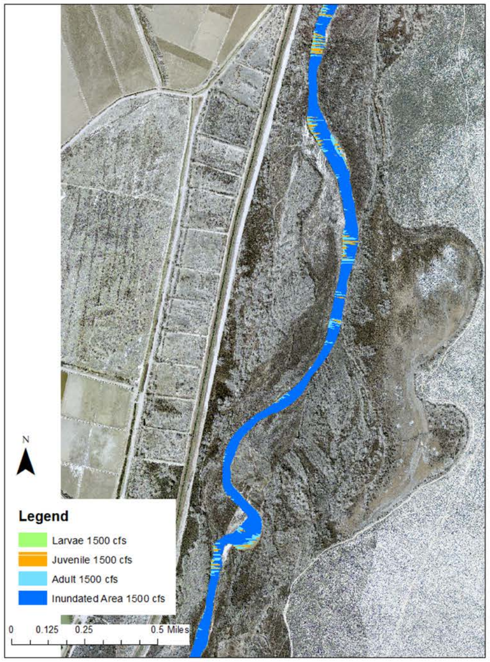
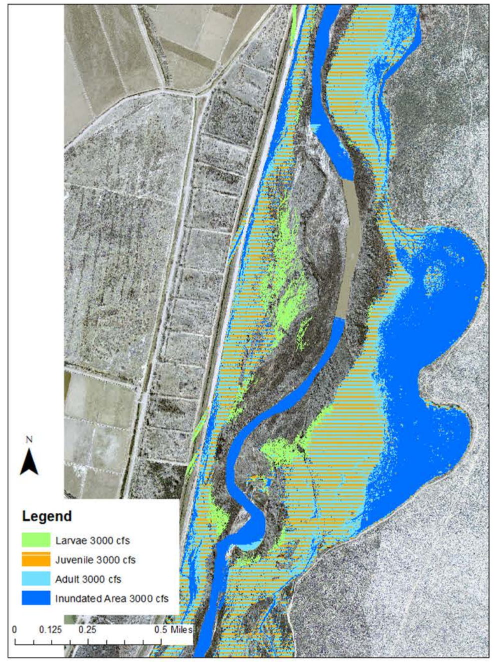



Figure A-35. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA3a (agg/deg 1516–1582).

B3.B Habitat (1500 cfs)




B3.B Habitat (3000 cfs)




B3.B Habitat (5000 cfs)

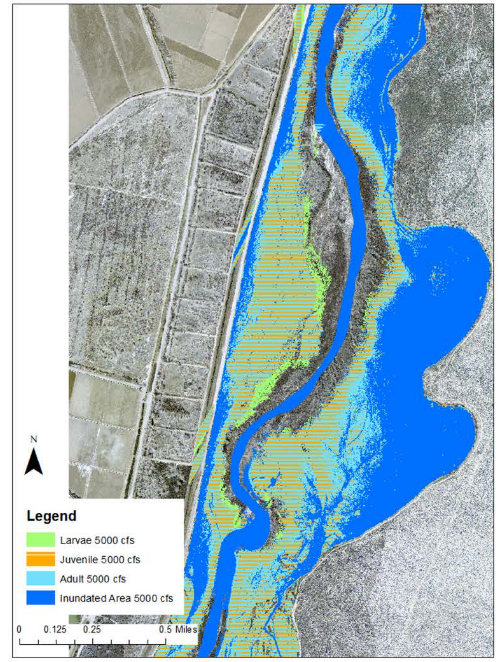



Figure A-36. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA3b (agg/deg 1516–1582).

B3.C Habitat (5000 cfs)

B3.C Habitat (3000 cfs)

B3.C Habitat (1500 cfs)

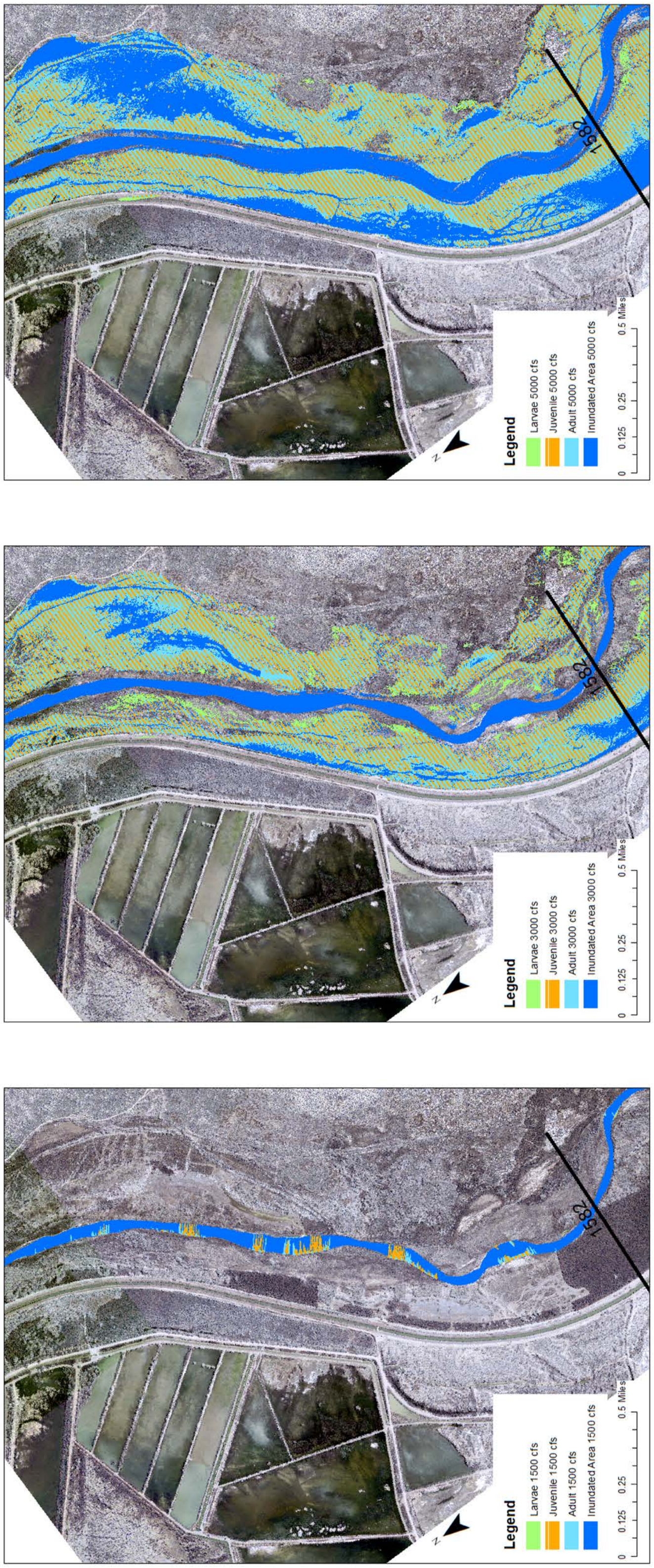



Figure A-37. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA3c (agg/deg 1516–1582).

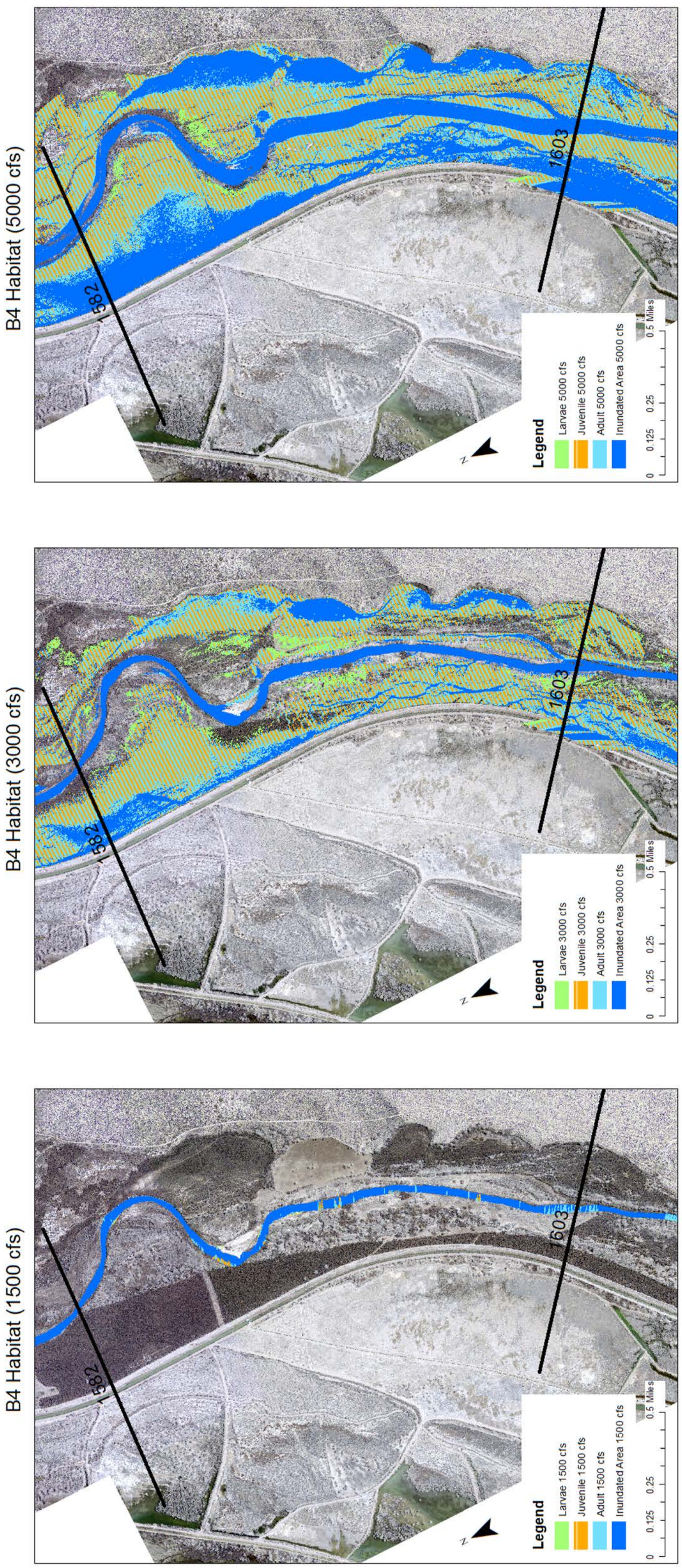



Figure A-38. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA4 (agg/deg 1582–1603).

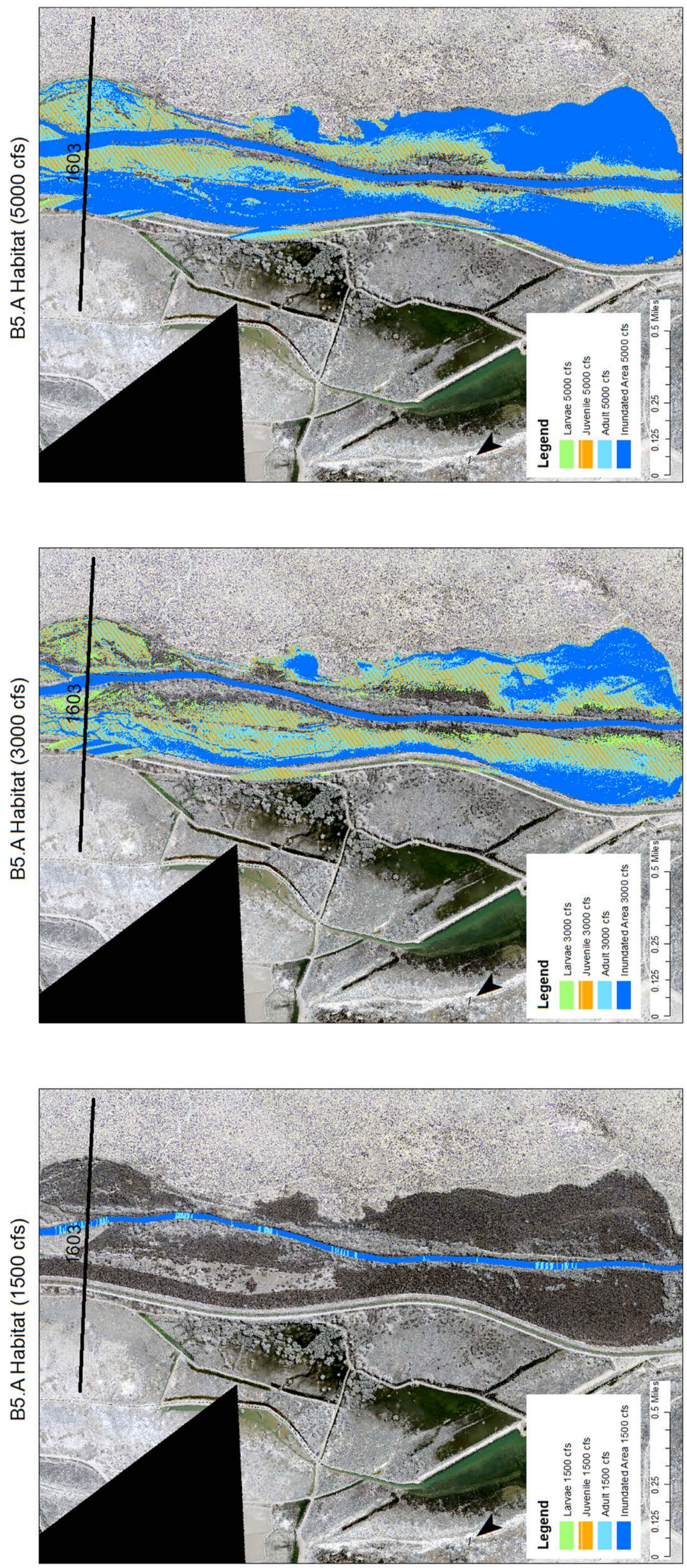
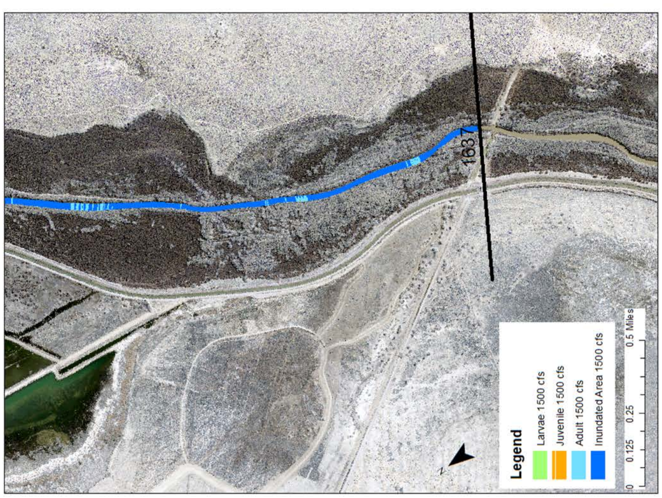
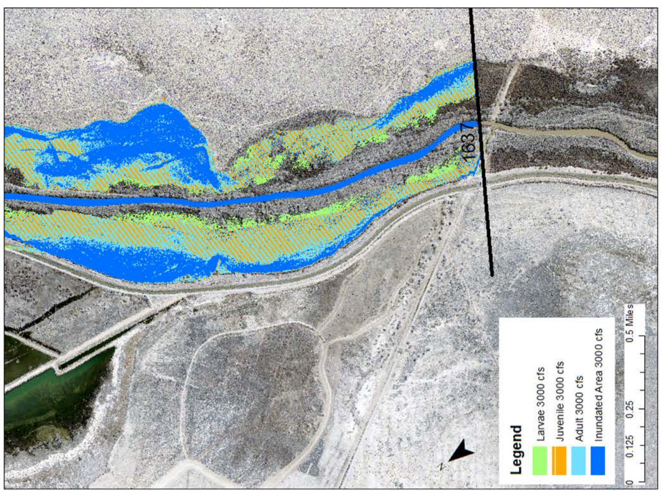



Figure A-39. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA5a (agg/deg 1603–1637).

B5.B Habitat (1500 cfs)




B5.B Habitat (3000 cfs)




B5.B Habitat (5000 cfs)

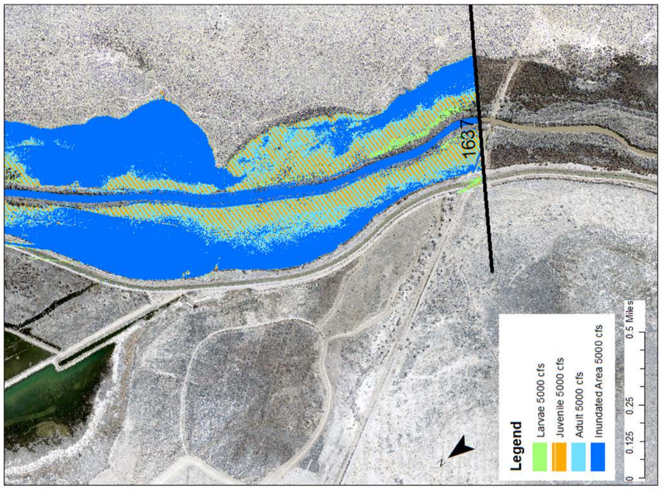



Figure A-40. Habitat mapping results obtained using RAS mapper (HEC-RAS) for subreach BDA5b (agg/deg 1603–1637).

# 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 began in 2017.

| Year | Month | | | | | | | | | | | | Sampling | Sampling Distribution | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| — | J | F | M | A | M | J | J | A | S | O* | N | D | Frequency | Angostura | Isleta | San Acaci | 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

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.

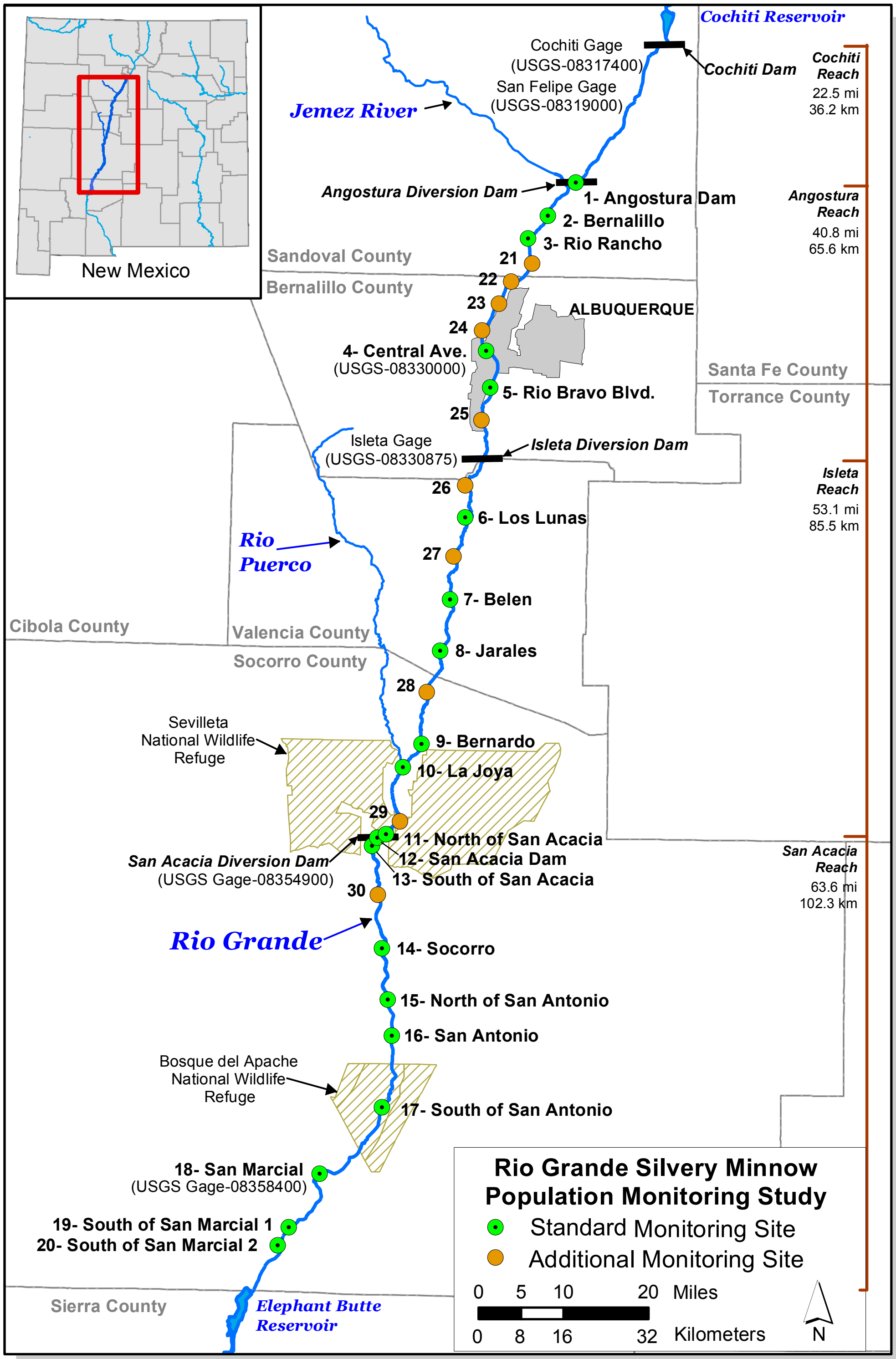



Figure B-1. Map of the study area for the Rio Grande Silvery Minnow Population Monitoring Program (from Dudley et al., 2020). Currently, 'standard' sites are sampled monthly April–November and 'additional' sites are sampled twice annually (April and October).

*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 ($\delta$), estimated lognormal density ($\mu$), and standard deviation of the estimated lognormal density ($\sigma$). 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 [$\delta$]) 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 $\delta$) 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 [$\delta$] vs. density [$\mu$]) 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 $\delta$, non-constant $\sigma$ in the lognormal distribution, or additional variation around $\delta$ and $\mu$ 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.

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# APPENDIX C
# SUPPLEMENTARY GEOMORPHIC DATA

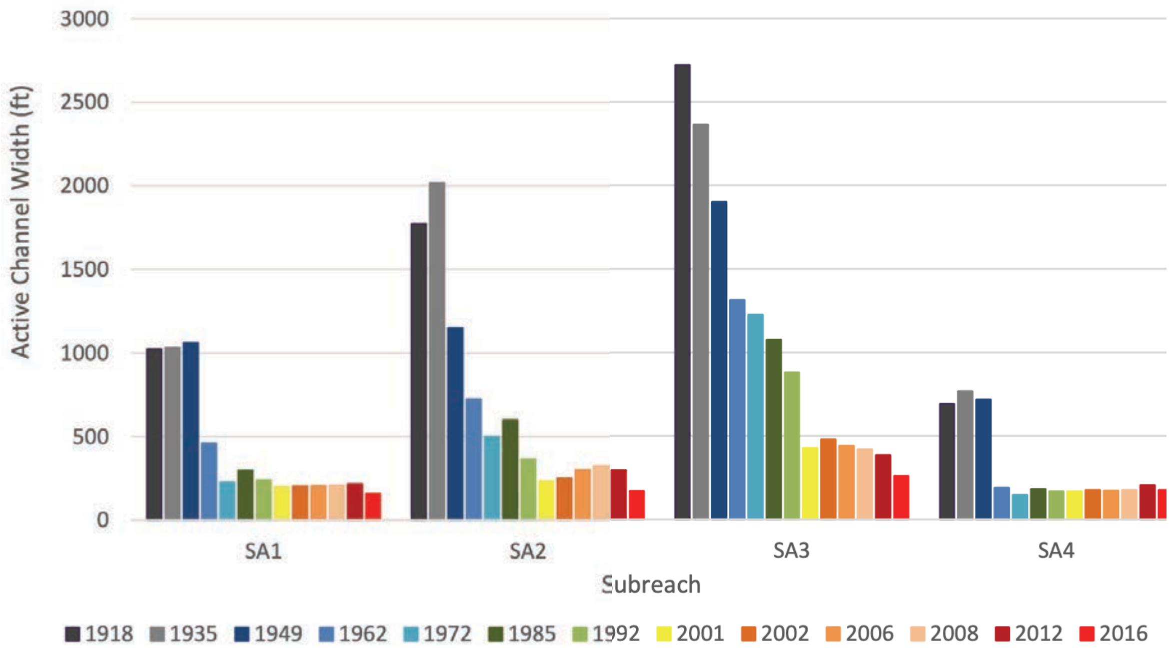


Figure C-1. Active channel width for subreaches SA1–SA4 1918–2016. From Doidge et al., (2020).

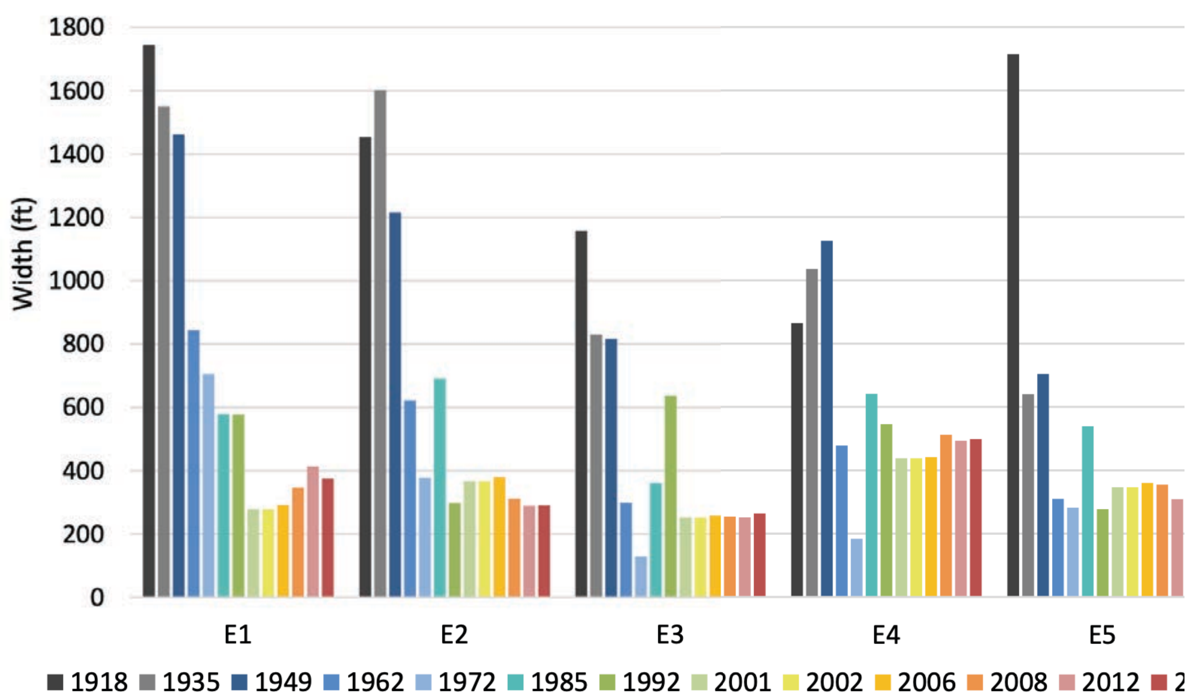


Figure C-2. Active channel width for subreaches E1–E5 1918–2016. From Beckwith and Julien (2020).

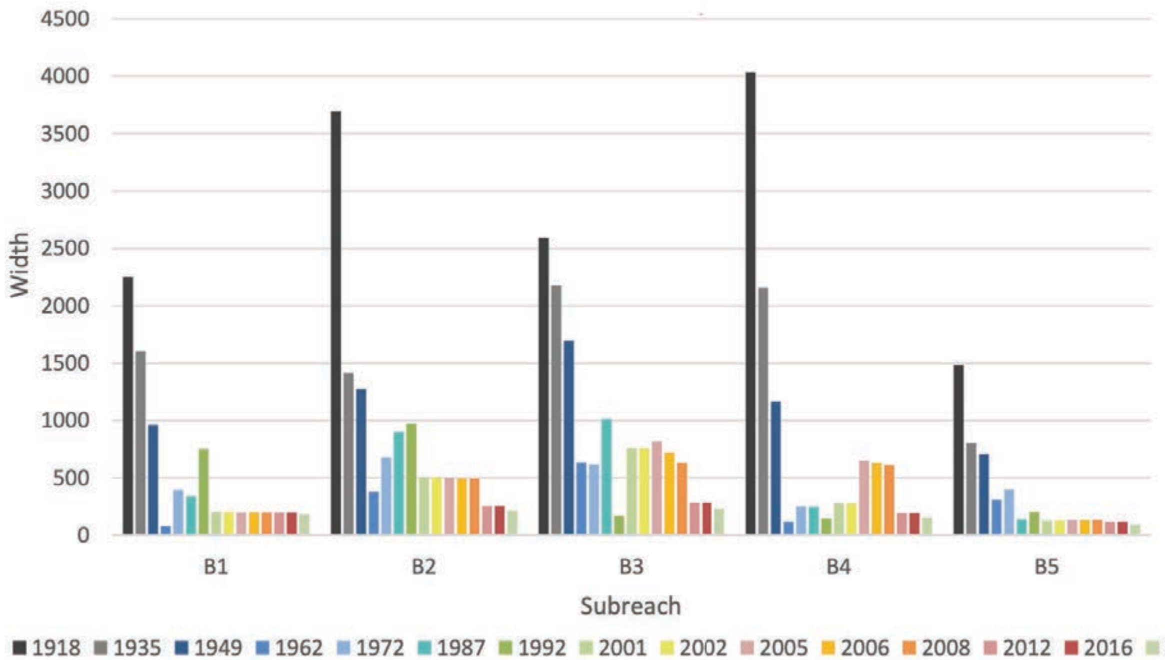


Figure C-3. Active channel width for subreaches BDA1–BDA5 1918–2019. From Schied et al., (2022).

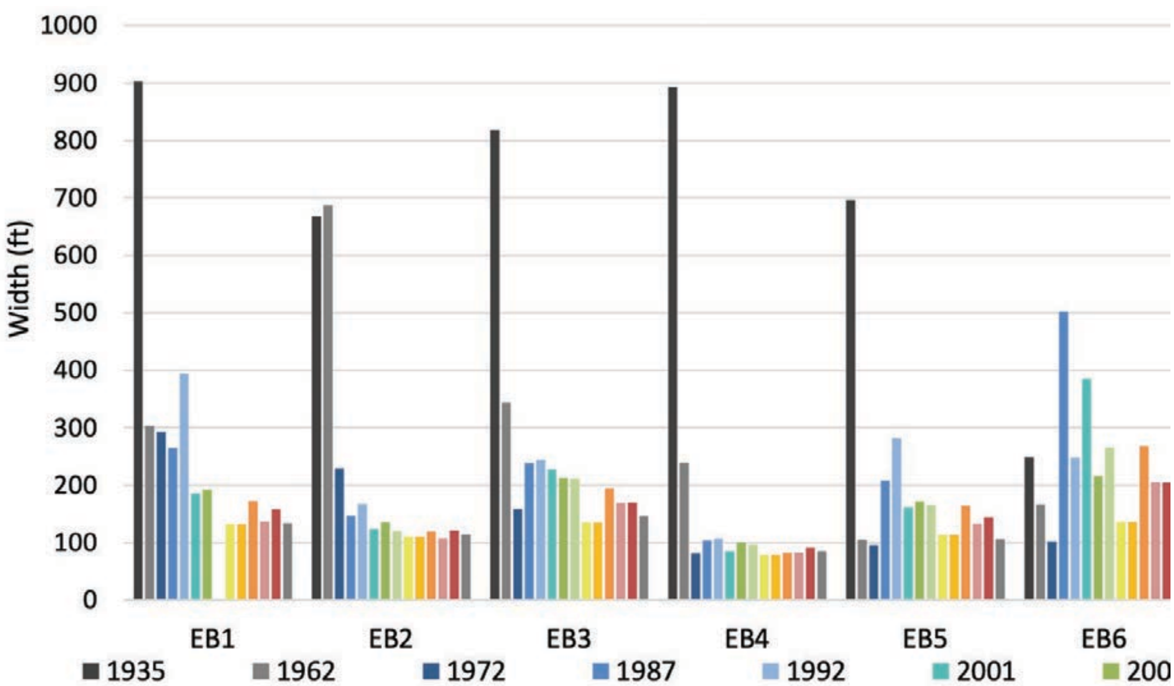


Figure C-4. Active channel width for subreaches EB1–EB6 1918–2019. From Sperry et al., (2020).

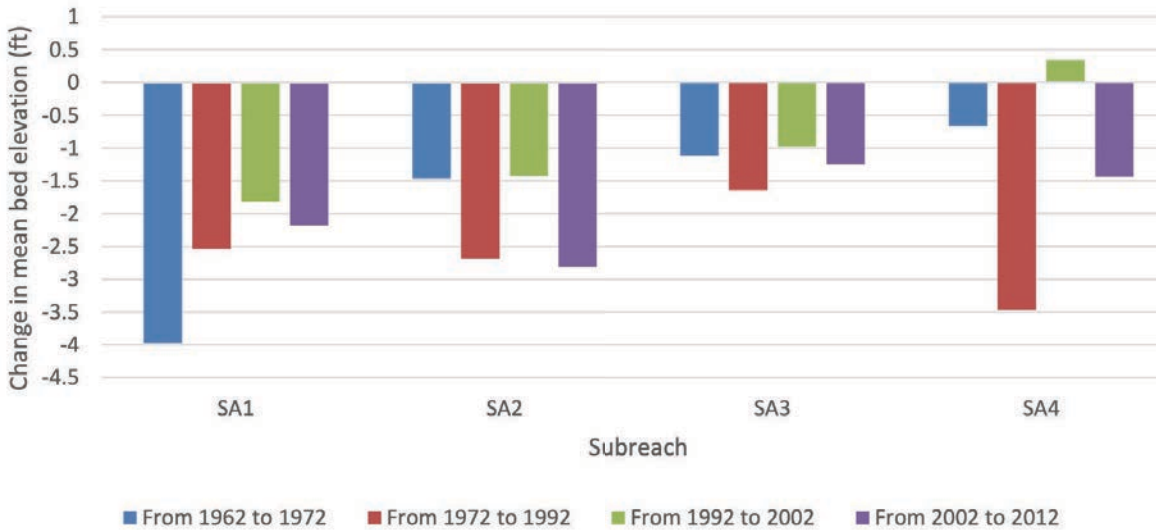


Figure C-5. Change in mean bed elevation for subreaches SA1–SA4 1962–2012. From Doidge et al., (2020).

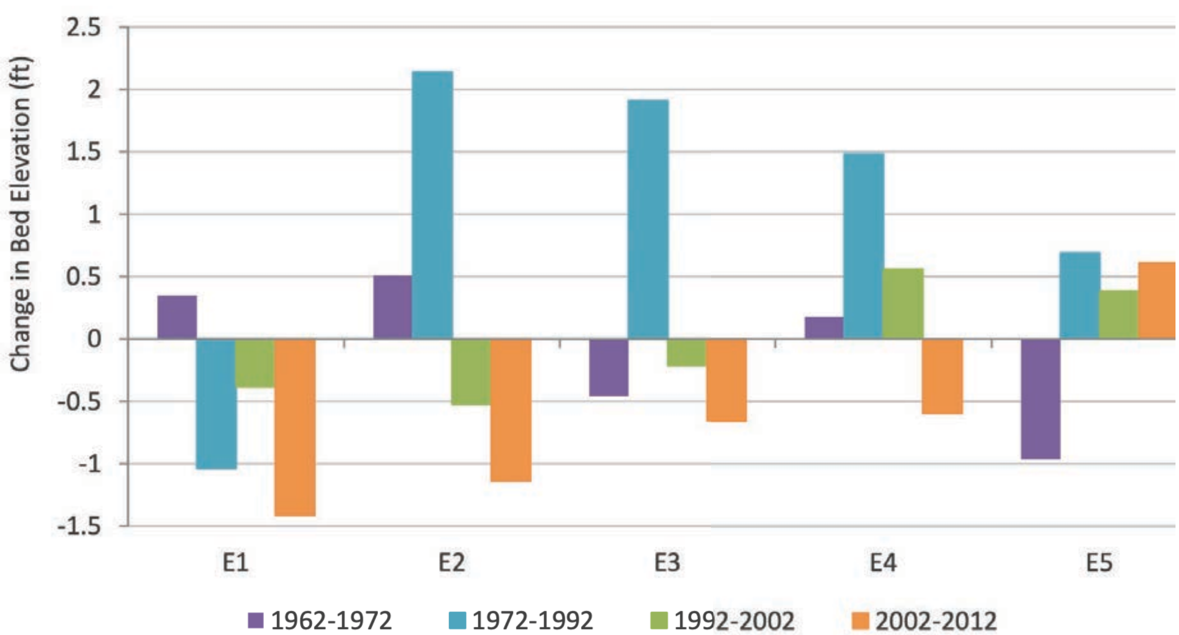


Figure C-6. Change in mean bed elevation for subreaches E1–E5 1962–2012. From Beckwith and Julien (2020).

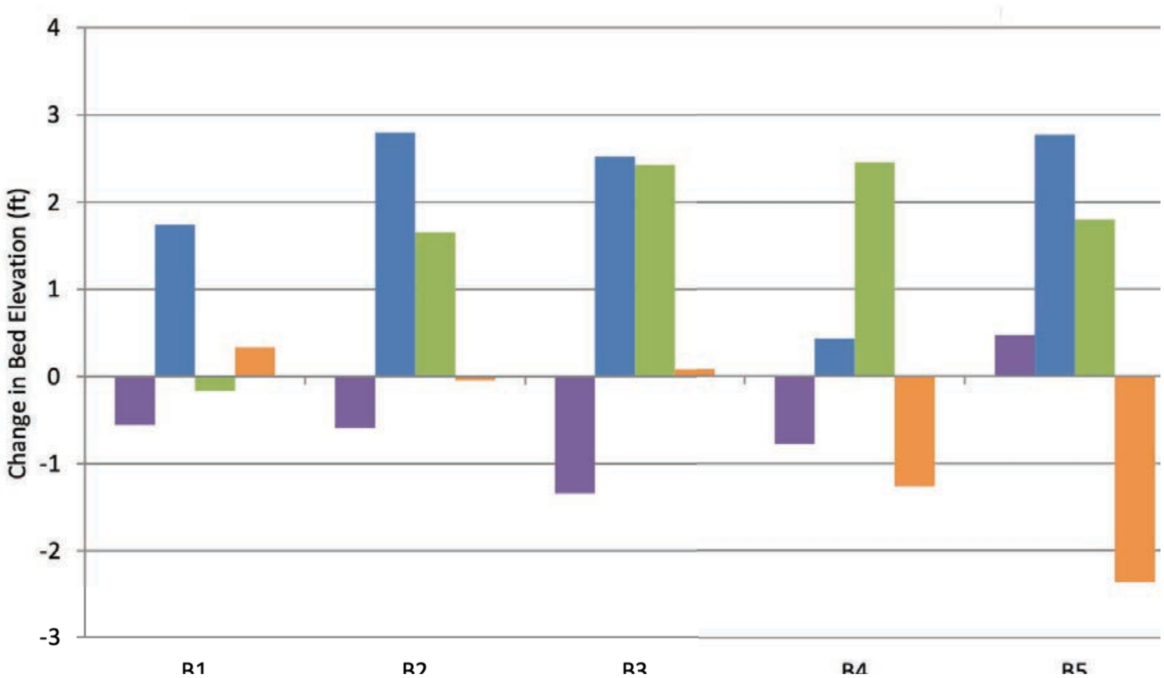


Figure C-7. Change in mean bed elevation for subreaches BDA1–BDA5 1962–2012. From Schied et al., (2022).

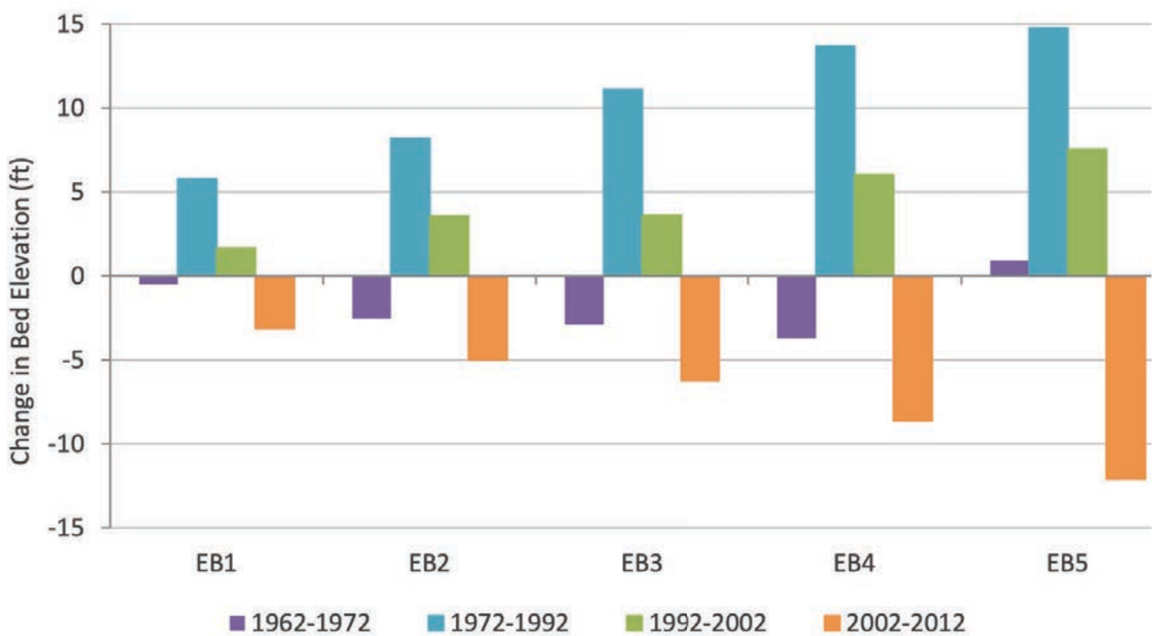


Figure C-8. Change in mean bed elevation for subreaches EB1–EB5 1962–2012. From Sperry et al., (2022).

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