Colorado's Water Supply Future

Colorado Water Conservation Board

Watershed Flow Evaluation Tool Pilot Study for Roaring Fork and Fountain Creek Watersheds and Site-Specific Quantification Pilot Study for Roaring Fork Watershed

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Section 1
Introduction

1.1 Nonconsumptive Needs Assessment Overview

In 2005, the Colorado legislature established the Water for the 21st Century Act. This act established the Interbasin Compact Process that provides a permanent forum for broad-based water discussions in the state. It created two new structures: (1) the Interbasin Compact Committee (IBCC), and (2) the Basin Roundtables. There are nine Basin Roundtables based on Colorado's eight major river basins and the Denver metro area as shown in Figure 1-1.

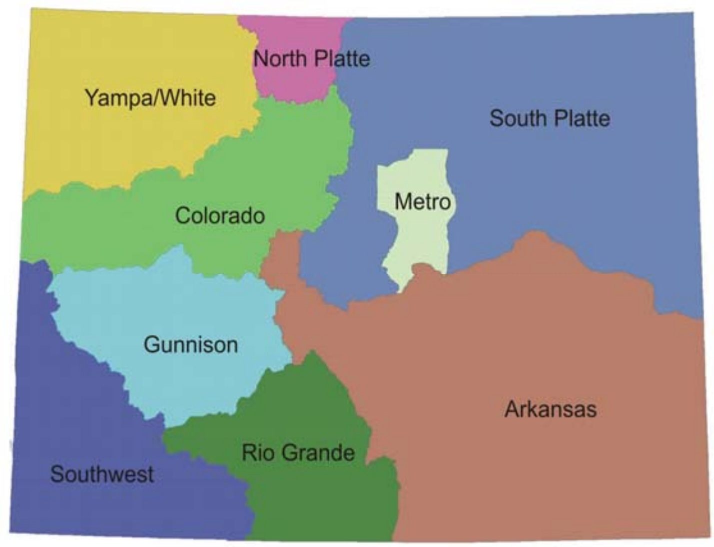


Figure 1-1 Colorado's Nine Basin Roundtables

As part of the Interbasin Compact Process, the Basin Roundtables are required to complete basinwide needs assessments. The needs assessments are to include the following:

- An assessment of consumptive water needs (municipal, industrial, and agricultural);
- An assessment of nonconsumptive water needs (environmental and recreational);
- An assessment of available water supplies (surface and groundwater) and an analysis of any unappropriated waters; and
- Proposed projects or methods to meet any identified water needs and achieve water supply sustainability over time.

The Watershed Flow Evaluation Tool (WFET) pilot study presented in this report is part of Phase I of the statewide technical assistance to the Basin Roundtables in completing their nonconsumptive needs assessments. Figure 1-2 shows the overview of the nonconsumptive needs assessment process. Phase I of the nonconsumptive needs assessment process focused on the following:

A. Expanding upon the existing set of environmental and recreational attribute maps that were developed through the Statewide Water Supply Initiative Phase 2 process, creating a statewide technical platform from which to build the nonconsumptive needs assessment process.

B. Identifying where environmental and recreational attributes are focused in the basins through a mapping processes conducted for each Basin Roundtable, and

C. Developing quantification tools that can be used for Phase II. These included:

   i. Completing a pilot of the WFET for Fountain Creek and Roaring Fork watersheds and

   ii. Completing a pilot of the site-specific quantification for the Roaring Fork watershed.

Phase II of the nonconsumptive needs assessments will include the following tasks:

- An examination of next steps for priority focus areas as directed by the Basin Roundtables;
- Flow evaluations as determined by the Basin Roundtables for focus areas as needed; and
- Basin Roundtable identification of projects and methods (both structural and nonstructural) to meet their identified nonconsumptive needs.

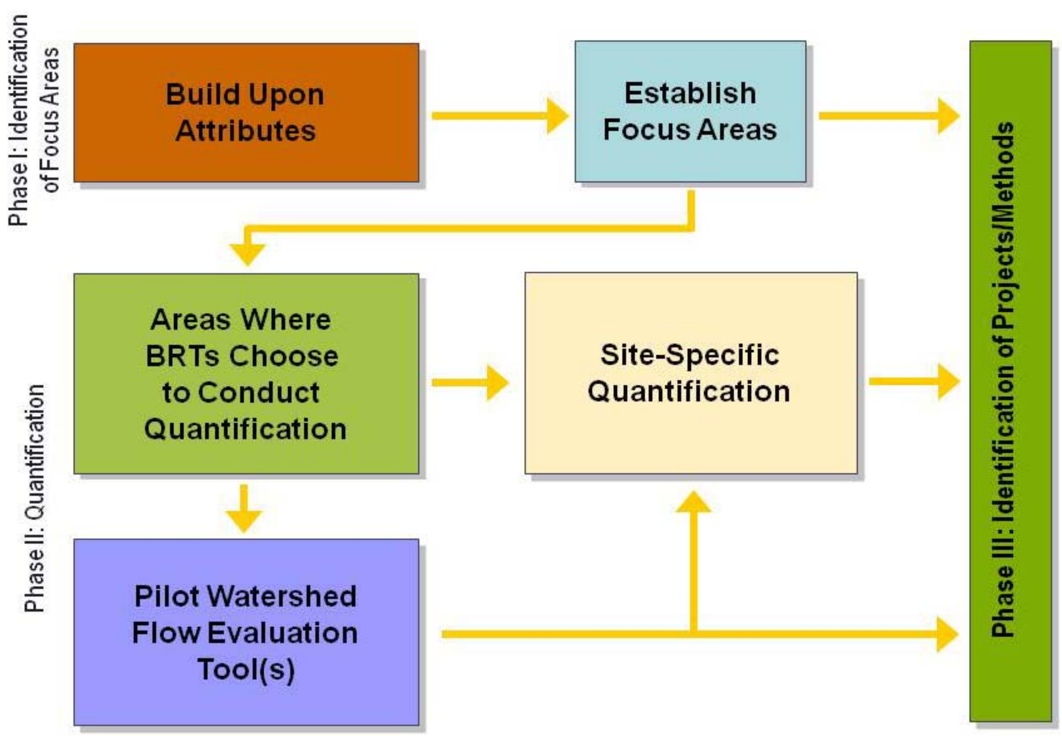


Figure 1-2 Overview of State of Colorado nonconsumptive needs assessment process

1.2 Purpose of Pilot Studies

Many of the Basin Roundtables have requested that the Colorado Water Conservation Board (CWCB) provide technical assistance in quantifying flow needs for the environmental and recreational priority areas that they have identified. The purpose of the *WFET pilot study* is to test the applicability of recent research developed by an international collaboration of researchers and water resource professionals at universities, government agencies, and non-governmental organizations (Poff et al. In Press and www.conserveonline.org/workspaces/eloha). The WFET provides a framework for examining ecological risk related to flow conditions at a watershed or regional level. The purpose of the *site-specific quantification* is to apply standard techniques in developing reach-based flow quantification based on historic data collection efforts. One of the main reasons research efforts have focused on regional assessments of flow conditions is that although there have been many site-specific flow quantifications completed around the U.S., the geographic extent of rivers and streams with site-specific quantification is still quite small.

Based on the findings of the WFET pilot studies and comparison with the site-specific study, the capabilities and limitations of the WFET are as follows:

Capabilities

- The WFET can provide a regional assessment of ecological risk conditions related to flow, identifying locations with minimal to high risk based on flow conditions for specific stream attributes without detailed site-specific information
- The WFET can identify areas that are at ecological risk based on flow conditions
- The WFET can provide a range of seasonal flow conditions that are associated with ecological risk levels
- The WFET can be used to target areas that need further site-specific studies
- The WFET is most suitable for use in areas with a detailed understanding of baseline and existing hydrologic conditions (i.e., areas where CWCB has developed a Decision Support System model)

Limitations

- The WFET is not intended to set flow prescriptions or rules for flow needs to the level of detail that would be required in a National Environmental Policy Act analysis or that might be needed to guide day-to-day management of a flow in a specific water project
- The WFET will not provide results as detailed or accurate as a site-specific analysis
- The WFET will not identify areas that are at ecological risk for factors not directly associated with flow conditions

The capabilities and limitations of the site-specific quantification techniques are as follows:

Capabilities

- Site-specific quantification can generate a lot information about a smaller geographic extent such as a river reach
- Site-specific quantification provides greater detail on multiple parameters than nonfield methods such as WFET
- Site-specific quantification directly relates channel characteristics to hydraulics, hydrology, and habitat
- Site-specific quantification can help validate the WFET results and refine risk level categories in the WFET

Limitations

- Site-specific quantification is based on data from short stream segments (hundreds of feet) and can be extrapolated only to relatively short segments (at most tens of miles) that the sample reach represents
- Site-specific quantification requires field data measured at the site, which leads to higher costs than desktop methods such as WFET
- Because of cost and time constraints, site-specific quantification is not appropriate for developing ecological risk levels on a regional scale
- The tools used during site-specific quantification were designed for analysis of fish habitat and were not specifically designed to address maintenance of other biological components (e.g., riparian plant communities) or physical attributes (sediment transport)

The WFET and site-specific methods are complementary of one another. For example, WFET results can identify areas where further site-specific studies need to be conducted. Also, historic site-specific studies can help validate and calibrate WFET results. Examples of validation and calibration will be discussed in Section 3 of this report.

The results of the WFET pilot studies in the Roaring Fork and Fountain Creek were considered in two ways. First, the results were examined based on whether the steps discussed in the research could be applied in Colorado in a manner that would provide meaningful technical results. Second, results were reviewed to provide recommendations for further application and refinement of the WFET. These findings and recommendations are discussed in Section 4 of this report.

1.3 Report Overview

Following is an overview of the report:

- Section 2 describes the methodology used to complete the WFET pilot for the Fountain Creek and Roaring Fork watersheds
- Section 3 describes the WFET pilot results for the Fountain Creek and Roaring Fork watersheds and compares Roaring Fork WFET results with the site-specific quantification for the Roaring Fork
- Section 4 discusses WFET pilot findings and recommendations
- Section 5 provides references for the report
- Appendix A is a detailed report describing the site-specific quantification that was completed for the Roaring Fork watershed
- Appendix B is a report detailing flow-ecology relationships for the State of Colorado completed by Colorado State University and used in the WFET pilot study
- Appendix C is the manuscript for "The Ecological Limits of Hydrologic Alteration (ELOHA): A New Framework for Developing Regional Environmental Flow Standards" from which the WFET approach is derived

Section 2
Watershed Flow Evaluation Tool Methodology

2.1 Watershed Flow Evaluation Tool Methodology Summary

The WFET methodology is based on the concept that a regional framework for understanding environmental and recreational flow needs is needed in Colorado. A key assumption of the methodology is that flow regime is a primary determinant of the structure and function of aquatic and riparian ecosystems for streams and rivers (Poff et al. 1997). Environmental flows are defined as "the term applied to explicit management of water flows through freshwater ecosystems such as streams, rivers, wetlands, estuaries and coastal zone to provide an appropriate volume and timing of water flow to sustain key environmental processes and ecosystem services valued by local communities" (Poff et al. In Press). Environmental flows include a variable flow regime versus a minimum low flow as shown in Figure 2-1. Figure 2-1 summarizes the different portions of the flow regime that are tied to ecological function. Low flows are needed to maintain aquatic habitat. Seasonal high flows are often needed to cue spawning of certain types of fish. Flood flows are needed to sustain riparian ecosystems, scour the channel, and to maintain alluvial storage (Postel and Richter 2004).

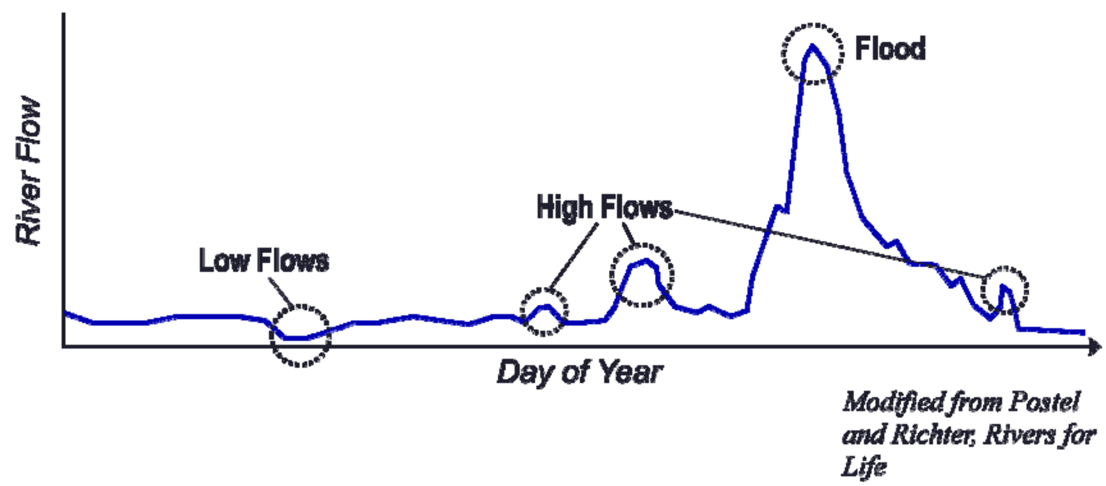


Figure 2-1 Flow Pattern and Its Relationship to Ecological Function

The steps used for the WFET pilot studies for the Fountain Creek watershed in the Arkansas Basin and the Roaring Fork watershed in the Colorado Basin are below:

- Step 1—Develop a hydrologic foundation
- Step 2—Calculate flow metrics and determine if changes have occurred based on water management in the watershed

- Step 3—Describe the quantitative relationship between important stream attributes (e.g., riparian forest) and key flow metrics (e.g., mean annual peak flow)
- Step 4—Develop ecological risk mapping that show areas that may be at risk due to changes in flow regime

Step 1—Develop A Hydrologic Foundation
Hydrologic data were gathered from U.S. Geological Survey (USGS) flow gages and modeled hydrology from CWCB's Colorado Decision Support System (CDSS) model for the Colorado River Basin. As part of developing the hydrologic foundation, two flow databases were constructed: one for baseline conditions (i.e., flow conditions that existed prior to today's system management), and one for existing flow conditions resulting from current system water management.

Step 2—Calculate Flow Metrics
As part of this step, The Nature Conservancy's Indicators of Hydrologic Alteration software (Richter et al. 1996) was used to calculate key flow metrics that relate to ecological conditions in the watersheds for the baseline and existing hydrologic databases. Flow metrics are statistics that summarize the key elements of a flow regime: magnitude, frequency, duration, timing, and rate of change.

Step 3—Develop Flow-Ecology Relationships
A literature search was conducted to link environmental and recreational attributes with flow metrics in Colorado. Quantitative relationships between stream attributes and those flow metrics calculated in step 2 were described. Appendix B contains the report describing these flow-ecology relationships.

Step 4—Develop Ecological Risk Mapping
Finally, results from steps 2 and 3 were used to develop mapping that related changes in flow from baseline to existing conditions to ecological risk.

Additional applications of the WFET are considered in Section 3.

The methodology for each of these steps for the Fountain Creek watershed and Roaring Fork watershed is described in the remainder of this section.

2.2 Hydrologic Foundation

This section describes the development of the hydrologic foundation for the Fountain Creek and Roaring Fork watersheds. The Fountain Creek hydrologic foundation was based on USGS data. The Roaring Fork hydrologic foundation was based on CDSS modeled data.

2.2.1 Fountain Creek Hydrologic Foundation

Long-term historical flow records from USGS gaging stations provided the hydrologic foundation for the Fountain Creek watershed pilot study presented here. Six gages, spatially distributed across the watershed, were used in this study (Figure 2-2 at the end of this section). Gage elevations ranged from approximately 6200 feet down to approximately 4700 feet, with a gradual shift from primarily snowpack-driven hydrology at the high elevation sites down to predominantly monsoonal hydrology in the summer at the lower gages. Primarily monthly data were used for this analysis.

Recorded flows are available only back to 1976 for USGS gage 07105500. Therefore, for the analysis presented here, data from two upstream tributary gages (07103700 and 07104000) were used to calculate a longer historical period of flows for this gage. Flows from the two upstream gages were combined and multiplied by area weighting factors to generate monthly flows for 07105500. The equation for this calculation is:

$$\mathbf{Q_{5500}} = (\mathbf{Q_{3700}} + \mathbf{Q_{4000}}) * \frac{\mathbf{A_{5500}}}{\mathbf{A_{3700+4000}}}$$

Where Q5500 = estimated flow at 07105500
Q3700 = measured flow at 07103700
Q4000 = measured flow at 07104000
A5500 = drainage area of 07105500
A3700 = drainage area of 07103700
A4000 = drainage area of 07104000

A close agreement was achieved between measured and calculated flows at this gage for the available overlapping period (Figure 2-3 at the end of this section).

A review of past studies, modeling, and the available gaging data indicates a significant change in watershed flow regimes occurring from approximately the 1970s to the early 1980s. For the downstream gages, this roughly corresponds to the commencement of the transbasin projects described above. This is reflected at the end of this section in Figure 2-4 which shows that based on analyses performed by the USGS (2000), a shift in the relationship between streamflow and precipitation occurred at USGS gage 07106500 in approximately 1980. After this time period, higher flow rates were observed for a given unit of precipitation, indicating additional sources of water in the system. Similar curves, developed by CDM, are shown in Figure 2-5 at the end of this section for USGS gages 07104000 and 07105500, our upper-most sites. The observed changes in these curves appear to be attributable to the development of new municipal wells in the upper portion of the watershed, as described above. The shift is more subtle for the downstream gage, 07105500, indicating that return flows from groundwater pumping in upper Monument Creek is diminished below the confluence with Fountain Creek.

Knowledge of specific water supply projects and the hydrologic analysis of gage data for Fountain Creek indicated the need to focus on the potential impacts of surface water augmentation in the watershed, primarily due to transbasin water delivery projects. The major transbasin water projects in the watershed include the Homestake, Blue River, and Fry-Ark projects. The Homestake project diverts water from the Eagle

River on the western slope and ultimately delivers this water into Rampart Reservoir (Figure 2-2) for use by the City of Colorado Springs. The Blue River project diverts from the Blue River on the western slope and delivers water to the North Catamount Reservoir (Figure 2-2) for use by the City of Colorado Springs. The Fry-Ark project diverts water from the Roaring Fork River basin on the western slope and delivers to the City of Colorado Springs and the City of Fountain via multiple reservoirs in the Arkansas River basin. Per the City of Colorado Springs, each of these major projects delivers water from reservoirs to the Colorado Springs and Fountain water treatment plants via pipelines rather than instream flow.

USGS gages 07104000 and 07105500 are both upstream of the primary point of municipal return flows (Las Vegas wastewater treatment plant). However, during the 1980s, the Town of Monument developed groundwater supplies as part of its long-term water supply portfolio. Municipal return flows from the Town of Monument occur at the Tri-Lakes Wastewater Treatment Plant (Figure 2-2).

Additionally, it is recognized that urbanization and resultant increases in impervious areas may also have contributed to observed changes in surface water flow regimes in the watershed.

2.2.2 Roaring Fork Hydrologic Foundation

The hydrologic foundation for Roaring Fork was based on the Upper Colorado River Basin Water Resources Planning Model (Upper Colorado River Model) that was utilized to generate the baseline (i.e., human influences removed) and existing conditions flows for the Roaring Fork River Basin. The Upper Colorado River Model, one of the water allocation models of the CDSS, was developed to simulate the availability of water to individual users and projects based on hydrology, water rights, and operating rules and practices in the Upper Colorado River Basin in which the Roaring Fork River is one of the major tributaries. The model uses nodes (representing reservoirs, major diversions, instream flow requirements, flow gages, etc.) and arcs (representing rivers, streams, channels, etc.) to construct the continuity in the system. Figure 2-6 at the end of this section shows the schematic of the Upper Colorado River Model. Figure 2-7 at the end of this section shows the distribution of the 47 nodes where hydrologic data was generated for the hydrologic foundation.

The Upper Colorado River Model is an implementation of the State of Colorado's Stream Simulation Model (StateMod), which is a program developed by the State of Colorado to simulate water allocation and accounting for making comparative analyses of various historic and future water management policies in a large-scale river basin. No modifications of the model were made for this study, and it was also assumed that the model output was sufficient for relative comparisons needed to complete the analysis of the changes between baseline and existing hydrologic conditions.

StateMod is capable of simulating both short-term (daily) and long-term (monthly) water allocation conditions. The latest versions of StateMod for daily and monthly simulation are versions 11.50 and 12.20, respectively. Time periods of the Upper Colorado River Model cover water years 1975 – 2005 (October 1, 1974, to September 30, 2005) for daily simulation and water years 1909 – 2005 (October 1908 to September 2005) for monthly simulation. More detailed information regarding StateMod and Upper Colorado River Model can be obtained in the CDSS website: http://cdss.state.co.us.

To generate baseline flow conditions, the current version (dated January 2007) of Upper Colorado River Model was used with the required changes to the model inputs to simulate the unimpaired flow conditions. These are the changes to the input files that CWCB uses to eliminate any human influences on the river basin. These changes to the input files turn off the diversions, instream flow rights, and reservoir operations in the basin. Both the daily and monthly model simulations were performed. Table 2-1 summarizes the inputs with associated changes.

Table 2-1 Summary of Upper Colorado River Model Inputs with Changes for Simulating Baseline Flow Conditions

Types of Simulation	Input Files	Changes
Daily	cmdlyB.rsp	Line 17, comment out cm2005.opr
	cmdly.ctl	Line 37, use 0 to represent the soil moisture accounting factor
	cm2005.ddr	Change every “on/off” from 1 to 0
	cm2005.ifr	Change every “on/off” from 1 to 0
	cm2005B.rer	Change every “on/off” from 1 to 0
Monthly	cm2005B.rsp	Line 17, comment out cm2005.opr
	cm2005.ctl	Line 37, use 0 to represent the soil moisture accounting factor
	cm2005.ddr	Change every “on/off” from 1 to 0
	cm2005.ifr	Change every “on/off” from 1 to 0
	cm2005B.rer	Change every “on/off” from 1 to 0

The current version (dated January 2007) of Upper Colorado River Model was used, without any changes in the inputs, to simulate the existing flow conditions. Both the daily (1975 – 2005) and monthly (1909 – 2005) model simulations were performed.

2.3 Flow Metric Calculation

Certain flow metrics can be considered ecologically important (Olden and Poff 2003). For both Fountain Creek and Roaring Fork watersheds the following flow metrics were determined to be relevant to one or more of the nonconsumptive needs assessment priority attributes and therefore were calculated at each node where flow data were available:

- Mean annual flow
- Mean August flow

- Mean September flow
- Mean January flow
- Mean annual peak daily flow

The Nature Conservancy's Indicators of Hydrologic Alteration software was used to calculate these flow metrics. For Fountain Creek, the baseline conditions period for the Fountain Creek analyses was designated as prior to 1970 while the existing conditions period was designated as subsequent to 1980. For the Roaring Fork watershed, the metrics were calculated for the baseline and existing conditions datasets outputs from the Upper Colorado River Model. These flow metrics were selected out of 67 statistical parameters (Richter et al. 1996) to accommodate the calculation of the ecologically relevant flow statistics described in Section 2.4. Because flow-ecology relationships do not capture all aspects of river health, maps that show the differences between baseline and existing conditions using the equation below were generated for each USGS gage or StateMod node. These results will be discussed in Section 3.

$$\frac{Q_{existing} - Q_{baseline}}{Q_{baseline}}$$

where Q=flow (cubic feet per second or cfs)

2.4 Flow-Ecology Relationships and Risk Mapping

Appendix B contains a detailed report regarding the review, synthesis, and analysis of literature to establish flow-ecology relationships for important environmental and recreational attributes for Colorado. The number of studies that were reviewed as part of this effort by stream type and community are summarized in Table 2-2. The report in Appendix B focuses on the areas highlighted in bold in Table 2-2.

Table 2-2 Number of Studies Reviewed for Establishing Flow-Ecology Relationships

Community Type or Attribute	Interior Western	Rocky Mountains	Great Plains	Total
Fish	**19**	**18**	**15**	52
Riparian vegetation	**20**	1	**8**	29
Invertebrates	9	**9**		18
Vertebrates (birds, beaver)	4			4
Terrestrial Invertebrates	2		1	3
Algae	2			2
Total	56	28	24	105

The remainder of this section is a discussion of how these ecology flow relationships were applied for the Fountain Creek and Roaring Fork watersheds to develop ecological risk maps.

2.4.1 Trout Flow-Ecology Relationships

To estimate the ecological risk for trout associated with late summer/early autumn low flows, the following equations from the flow-ecology relationships developed for trout were used to link ecological risk categories for baseline and existing hydrologic conditions for trout in the Roaring Fork and Fountain Creek watersheds:

Existing Conditions:

$$\frac{(\text{Mean August } Q_{\text{existing}} + \text{Mean September } Q_{\text{existing}}) \div 2}{\text{Mean Annual } Q_{\text{baseline}}}$$

Baseline Conditions:

$$\frac{(\text{Mean August } Q_{\text{baseline}} + \text{Mean September } Q_{\text{baseline}}) \div 2}{\text{Mean Annaul } Q_{\text{baseline}}}$$

where:

Q=flow (cfs)

These equations derive the percentage of mean annual flow that occurs during the low flow summer months. A GIS was used to assign a color representing the estimated ecological risk to each USGS gage (Fountain Creek) or StateMod node (Roaring Fork). The node colors and differentiation among risk levels were derived directly from the flow-ecology relationships for trout described on page 24 of Appendix B and are as follows:

- <10%: Red or inadequate to support trout (high ecological risk)
- 10 – 15%: Orange or potential for trout support is sporadic (significant ecological risk)
- 16 – 25%: Yellow or may severely limit trout stock every few years (moderate ecological risk)
- 26 – 55%: Green or low flow may occasionally limit trout numbers (minimal ecological risk)
- >55%: Blue or low flow may very seldom limit trout (low ecological risk)

2.4.2 Fountain Creek Warm Water Fishes

The warm water fishes found in the Fountain Creek watershed include Arkansas Darter and other plains minnows. To estimate the ecological risk for warm water fishes associated with late summer/early autumn low flows, the following equations from the flow-ecology relationships developed for warm water fishes were used to relate ecological risk categories to baseline and existing flow conditions for warm water fishes in the Fountain Creek watershed:

Existing Conditions:

$$\frac{(\text{Mean August } Q_{\text{existing}} + \text{Mean September } Q_{\text{existing}}) \div 2}{\text{Mean Annual } Q_{\text{baseline}}}$$

Baseline Conditions:

$$\frac{(\text{Mean August } Q_{\text{baseline}} + \text{Mean September } Q_{\text{baseline}}) \div 2}{\text{Mean Annaul } Q_{\text{baseline}}}$$

where:

Q=flow (cfs)

These equations derive the percentage of mean annual flow that occurs during the low flow summer months. An ecological risk category was assigned to each USGS gage location. A GIS was used to assign a color representing ecological risk. The node colors and differentiation among risk levels were based on the flow-ecology relationships for warm water fishes described in Appendix B (pages 31 – 34) and were simplified as follows:

- <10%: Red or severe degradation to warm water fishes (high ecological risk)
- 10 – 30%: Orange or poor or minimum habitat for warm water fishes (moderate ecological risk)
- 31 – 40%: Yellow or fair or degrading habitat for warm water fishes (minimal ecological risk)
- >40%: Green or good habitat for warm water fishes (low ecological risk)

2.4.3 Roaring Fork Warm Water Fishes

The warm water fishes found in the Roaring Fork watershed include Bluehead Sucker and Flannelmouth Sucker. To estimate the ecological risk for warm water fishes associated with late summer/early autumn low flows, the following equations from the flow-ecology relationships developed for warm water fishes were used to relate ecological risk categories to baseline and existing flow conditions for warm water fishes in the Roaring Fork watershed:

$$\frac{\left(\left(0.452\log_{10}\left(\frac{\text{AugQ}_{\text{existing}} + \text{SepQ}_{\text{existing}}}{2}\right) - 0.51\right) - \left(0.452\log_{10}\left(\frac{\text{AugQ}_{\text{baseline}} + \text{SepQ}_{\text{baseline}}}{2}\right) - 0.51\right)\right)}{\left(0.452\log_{10}\left(\frac{\text{August Mean } Q_{\text{baseline}} + \text{September Mean } Q_{\text{baseline}}}{2}\right) - 0.51\right)}$$

where:

Q=mean monthly flow (cfs)

The above equation is based on the flow-ecology relationship for Flannelmouth Sucker from Appendix B (pages 42 – 43) and shown in Figure 2-8 below. This equation estimates biomass based on the flow-ecology relationship (Figure 2-8) for baseline and existing conditions and assesses the percent change in biomass from baseline to existing conditions.

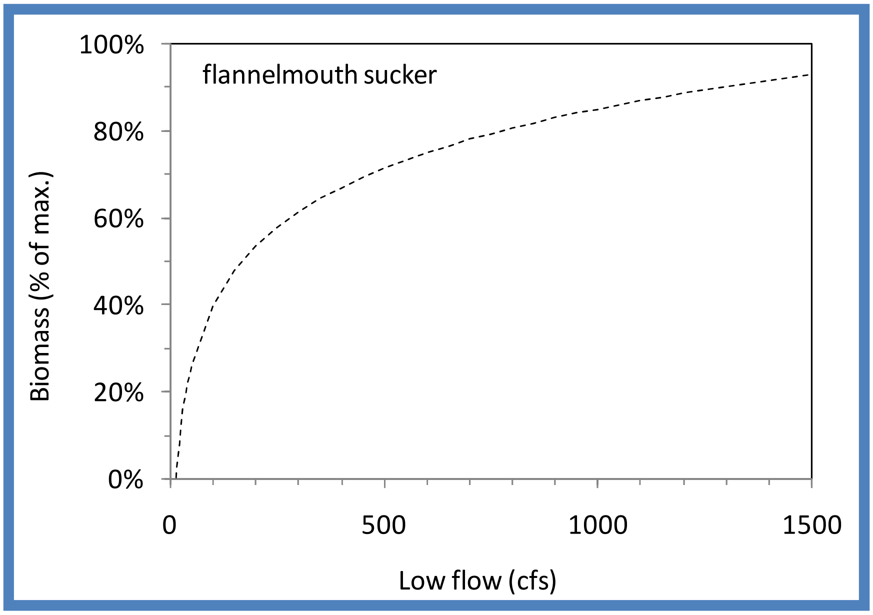


Figure 2-8 Flow-Ecology Relationship for Flannelmouth Sucker

The node colors and differentiation among risk levels were derived directly from the flow-ecology relationships for Flannelmouth Sucker. The risk levels based on biological expertise are as follows:

- <10%: Blue or low ecological risk based on potential biomass reduction
- 10 – 25%: Green or minimal ecological risk based on potential biomass reduction
- 26 – 50%: Yellow or moderate ecological risk based on potential biomass reduction
- ≥51%: Red or high ecological risk based on potential biomass reduction

2.4.4 Fountain Creek Erosion Potential

The flow-ecology relationships presented above and in Appendix B for Fountain Creek have a substantial degree of uncertainty because (a) data supporting these relationships are derived from instances of stream depletion, not augmentation, and (b) the geomorphic processes in Fountain Creek have changed due to flow augmentation, and these changes likely affect flow-ecology relationships in Fountain Creek. Given the primary influence of geomorphic processes on flow-ecology relationships in Fountain Creek, a preliminary examination was conducted of how sediment transport capacity and long-term erosion potential downstream of Colorado Springs have been influenced by flow regime changes occurring after 1980. The combined effects of flow and sediment regime on Fountain Creek were examined using Magnitude-Frequency Analysis (MFA; Wolman and Miller, 1960). In this approach, the estimated geomorphic effectiveness (i.e., long-term sediment transport) of different flow levels is multiplied by the likelihood of occurrence (Pickup and Warner, 1976; Andrews, 1980).

Fountain Creek below its confluence with Monument Creek is primarily an alluvial sand bed stream with continuous sediment transport and a diverse mosaic of channel forms created by temporal sequences of flow and sediment supply. Aquatic life and riparian attributes of Fountain Creek are inextricably linked with geomorphic processes. Along the mainstem of Fountain Creek, the quality, the quantity, and spatial distribution of instream habitats are controlled by the interaction between streamflow and geomorphic setting. Similarly, the condition and functioning of the riparian corridor depend on channel forms and floodplain connectivity during high flow events. Ongoing erosion and accelerated channel adjustment processes in lower Fountain Creek have been well-documented in previous studies (URS, 2007). Such rapid and complex channel dynamics require the development of flow-ecology linkages within specific geomorphic contexts, at a spatial resolution finer than that used in most watershed scale flow evaluation tools (Poff et al. In Press).

In practical applications of MFA, discharge values are typically arranged into a specified number of discrete classes, referred to henceforth as *bins*. The number of observations in each bin represents a flow frequency relative to the total number of flows recorded. The product of the sediment transport capacity of a representative flow from each bin and its flow frequency produces an estimate of how much sediment is transported by each bin. This procedure results in a series of discrete product values that form an effectiveness curve, with the effective discharge (Qeff) being the flow corresponding to the maximum. The area under the effectiveness curve estimates the time-integrated sediment load transported through the channel.

MFA was performed using USGS streamflow data from the Fountain and Pueblo gages on the Fountain Creek mainstem. At-a-station hydraulic geometry characteristics, grain size distributions, and flow resistance information for segments proximate to the gages were compiled from previous studies (URS, 2007). The GeoTools software package (Bledsoe et al., 2007) was used to perform the MFA computations with the Brownlie (1981) and Wilcock and Kenworthy (2002) sediment transport relationships using both 25 and 30 logarithmic bins. The results of the erosion potential analysis are described in Section 3.

2.4.5 Roaring Fork Recreation

The first step in estimating the recreational risk levels was to define the recreational season for kayaking and rafting in the Roaring Fork watershed. This was completed using two methods. First, recreational permitting entities such as the U.S. Forest Service and Pitkin County were contacted to establish what portion of the year they issue permits. Second, historical flow data were compared to guidebooks for the region and the beginning of the recreation season was set equal to the month when historical flow was met or exceeded flows recommended for a given reach in the guidebooks (Banks and Eckhardt 1999 and Stafford and McCutchen 2007). Based on this analysis, the recreation season for the Roaring Fork watershed for kayaking and rafting was considered May through August except for the very lower part of the watershed where permits for rafting occur year round.

After the recreation season was defined, the Alberta equation—shown in Figure 2-9 below and described in more detail in Appendix B (pages 34 – 35)—was used to estimate flow conditions that would be needed to support a minimum and preferred recreational experience at each node in the recreational reaches. The relationships in Figure 2-9 were based on paddler surveys, stage-discharge modeling and expert judgment from guide books.

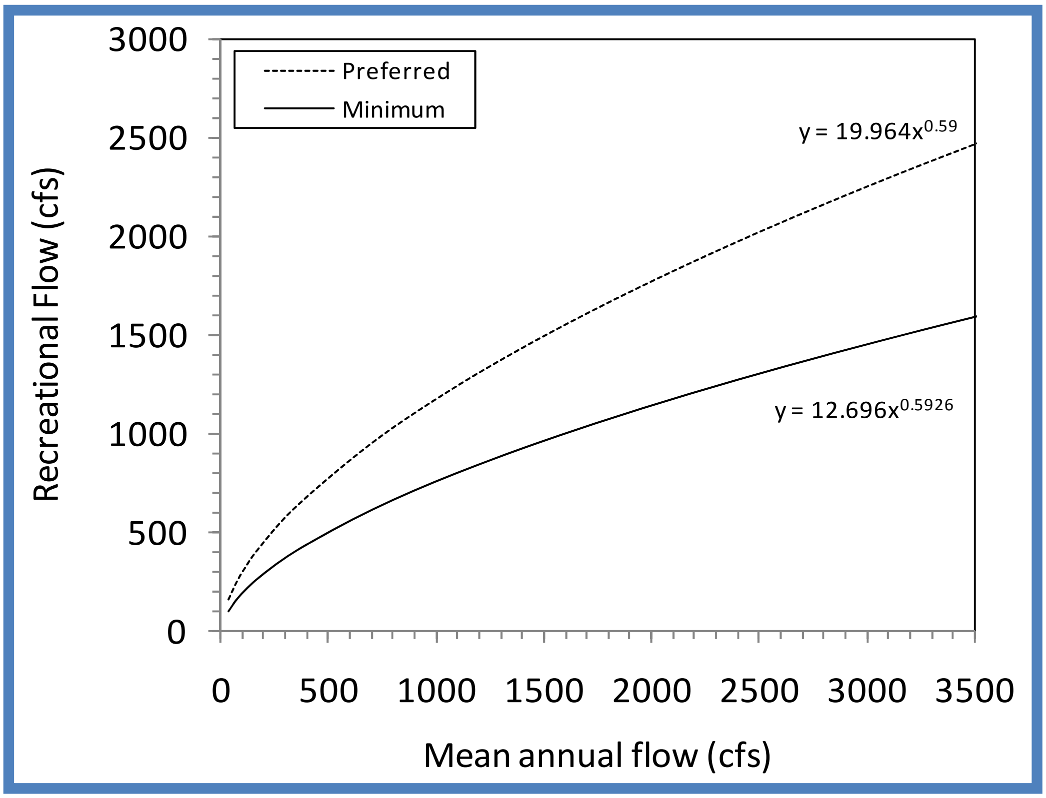


Figure 2-9 The Alberta Equation

To assess whether flow conditions estimated from Alberta equation were applicable to the Roaring Fork watershed, the flows generated from the equations were compared to American Whitewater flow recommendations (www.americanwhitewater.org), guidebook flow recommendations, and recommended recreation flows from the State of the Roaring Fork Watershed Report (Clarke et al., 2008).

The summary of this information is presented in Table 2-3. This table shows runs described from the reference materials, the USGS gage that the recommend flows are based upon, and the recommended flows from the reference materials.

Table 2-3 Summary of Recommended Recreation Flows

Run	Reference Gage	Recommended Flows (cfs) Stafford & McCutchen (2007)	Banks and Eckhart (1999)	State of the Watershed Report
Crystal Drainage				
Avalanche Down	Crystal above Avalanche Creek	500 – 700	>500	600 – 4000
Catherine Store	Roaring Fork at Emma	>800	not in guidebook	not specified
Narrows and Meatgrinder	Crystal above Avalanche Creek	350 – 1100	500 – 1200	not specified
Bogan Canyon	Crystal above Avalanche Creek	400 – 1200	not in guidebook	not specified
Yule Creek	Crystal above Avalanche Creek	400 – 1600	not in guidebook	not specified
Crystal Gorge	Crystal above Avalanche Creek	200 – 500	350	600 – 4000
South Fork Crystal	Crystal above Avalanche Creek	400 – 1200	not in guidebook	600 – 4000
Crystal Mill Falls	Crystal above Avalanche Creek	500 – 1500	>1500	600 – 4000
North Fork Crystal	Crystal above Avalanche Creek	350 – 1200	not in guidebook	600 – 4000
Upper Yule	Crystal above Avalanche Creek	700 – 1500	not in guidebook	not specified
Muddy Creek	Crystal above Avalanche Creek	200 – 400	not in guidebook	not specified
Fryingpan Drainage				
North Fork Fryingpan	No recommendations	No recommendations	not in guidebook	not specified
Upper Fryingpan	Fryingpan near Thomasville	300 – 600	300 – 1000	300 – 1500
Lower Fryingpan	Fryingpan near Ruedi	300 – 800	800	300 – 1500
Lime Creek	No recommendations	150	not in guidebook	not specified
Roaring Fork Drainage				
Cemetery	Roaring Fork at Glenwood	500 – 800	Not specified	1000 – 6000

Table 2-3 Summary of Recommended Recreation Flows

Run	Reference Gage	Recommended Flows (cfs)		
		Stafford & McCutchen (2007)	Banks and Eckhart (1999)	State of the Watershed Report
Upper Roaring Fork	Roaring Fork above Difficult Creek	150 – 450	Not specified	600 – 4000
Castle Creek	Roaring Fork at Maroon Creek	450 – 800	1000 – 2000	not specified
Grottos Park and Huck	No recommendations	No recommendations	Not in guidebook	not specified
Slaughterhouse	Roaring Fork at Maroon Creek	450 – 2500	700 – 2700	600 – 1800
Upper Woody Creek	Roaring Fork at Maroon Creek	500 – 1500	500 – 2000	not specified
Lower Woody Creek	Roaring Fork at Maroon Creek	500 – 1500	1700	not specified
Toothache Rapid	No recommendations	No recommendations	No recommendations	not specified

Table 2-4 shows the preferred and recommended flows based on the Alberta equation summarized in Figure 2-9 for each reference USGS gage in Table 3. The baseline mean annual flows from the Upper Colorado River Model for each reference gage were used for estimating the preferred and minimum flows in table 2-4.

The estimated preferred and minimum flows that were calculated using the baseline flow conditions in Table 2-4 were compared with Table 2-3. This comparison indicates that the estimated flows using the Alberta equation approximate the low end of the range of the reference flows for the majority of the recreation runs presented in Table 2-3.

Table 2-4 Preferred and Minimum Flows from the Alberta Equation by Reference Gage

Reference Gage	Preferred Flow (cfs)	Minimum Flow (cfs)
Crystal above Avalanche Creek	600	400
Roaring Fork at Emma	1500	900
Roaring Fork at Glenwood	1500	900
Roaring Fork above Difficult Creek	400	200
Roaring Fork at Maroon Creek	700	400
Fryingpan near Thomasville	400	300
Fryingpan near Ruedi	500	300

Because the estimated flows from the Alberta equation approximated the low range of the reference flows, the following approach was utilized to estimate risk levels for recreation:

- The preferred and minimum flows were calculated for both baseline and existing conditions using the following equations:

  $$\text{Preferred Flow Condition} = 19.965Q^{0.59}$$

  $$\text{Minimum Flow Condition} = 12.696Q^{0.5926}$$

  where: Q=flow (cfs)

- To estimate risk levels for recreation, the calculated minimum and preferred flows were compared to the mean monthly flow for each month in the defined recreation season for baseline and existing conditions.

- If the mean monthly flow met or exceeded the calculated minimum or preferred flows then the month was counted as being suitable for recreation for baseline and existing conditions.

- For baseline and existing conditions, the total number of months for the recreation season counted as suitable was mapped for each node based on the following risk levels:

  0—Red or no months considered suitable for recreation (high risk)

  1—Orange or one month considered suitable for recreation (significant risk)

  2—Yellow or two months considered suitable for recreation (moderate risk)

  3—Green or three months considered suitable for recreation (minimal risk)

  4—Blue or four months considered suitable for recreation (low risk)

2.4.6 Roaring Fork Riparian

To estimate the ecological risk for riparian areas associated with reduced high flow, the following equation from the flow-ecology relationships developed for riparian was used to calculate the ecological risk for riparian areas in the Roaring Fork watersheds:

$$\frac{\text{Annual Peak Daily Flow}_{\text{existing}} - \text{Annual Peak Daily Flow}_{\text{baseline}}}{\text{Annual Peak Daily Flow}_{\text{baseline}}} \text{x}1.18$$

This equation was derived from the following flow-ecology relationship discussed in Appendix B (pages 11 – 13) and shown in Figure 2-10 below. The flow-ecology relationship is based on difference in peak flow magnitude from existing and baseline conditions.

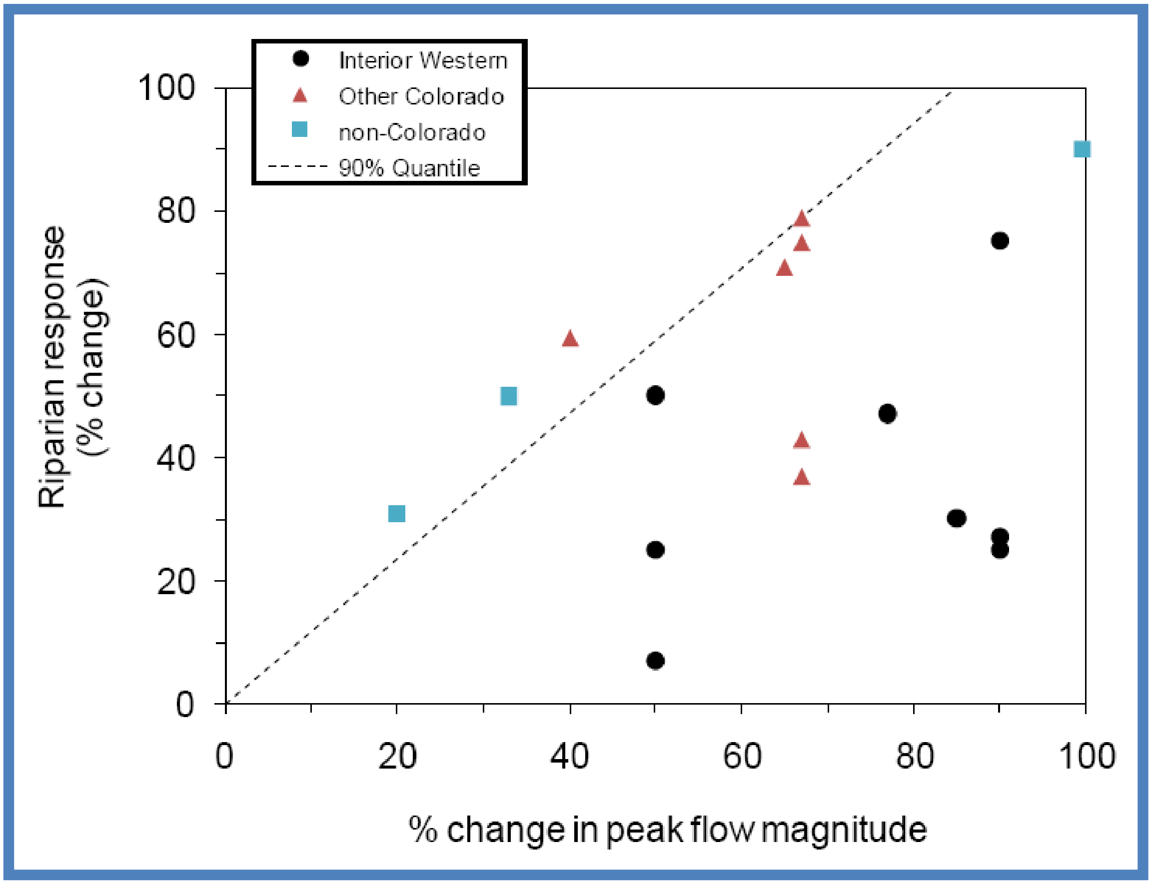


Figure 2-10 Flow-Ecology Relationship for Riparian Areas

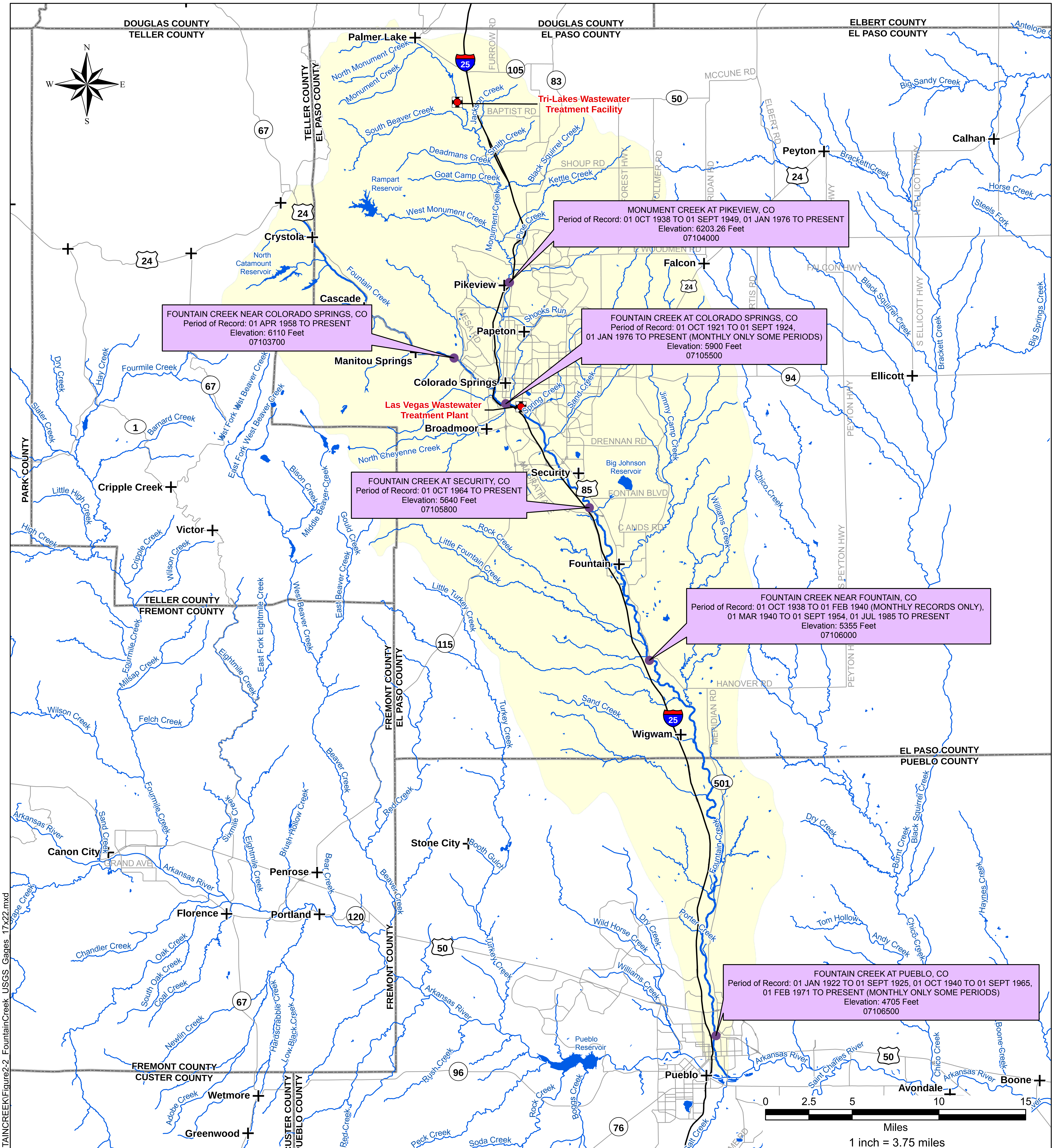
The riparian flow-ecology relationship curve was heavily influenced by research on narrowleaf cottonwood, which has an upper elevation limit of 9600 feet (Carsey et al. 2003), so the riparian ecological risk was calculated only for nodes and reaches below this elevation. Using a GIS, each node was assigned a color based on the ecological risk. The node colors and differentiation among risk level for the mapping are as follows:

- <10%: Green or low risk for riparian change
- 10 – 25%: Yellow or minimal risk for riparian change
- 25 – 50%: Orange or moderate risk for riparian change
- >50%: Red or high risk for riparian change

2.4.7 Mapping of Stream Reaches between USGS Gages and StateMod Nodes

For a subset of the maps generated for Fountain Creek and the Roaring Fork inferences of ecological risk categories between nodes were inferred based on consistent criteria. The following logic was used to assign ecological risk between USGS gages or StateMod nodes:

- Tributaries with known diversions above nodes have been assigned a conditional ecological risk of the downstream node. This is displayed on the mapping by using a dashed colored line that is the same as the downstream node.
- Tributaries without information about diversions or without node information have not been assigned an ecological risk level.
- Mainstem reaches above major diversions without node information above the diversion have not been assigned an ecological risk level.
- Mainstem reaches downstream of nodes were assigned the ecological risk of the upstream node to the next downstream node.



Legend

- USGS Stream Gages
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 2-2
Fountain Creek Watershed
Flow Evaluation Tool

USGS Gage and Wastewater Treatment Plant Locations

DRAFT

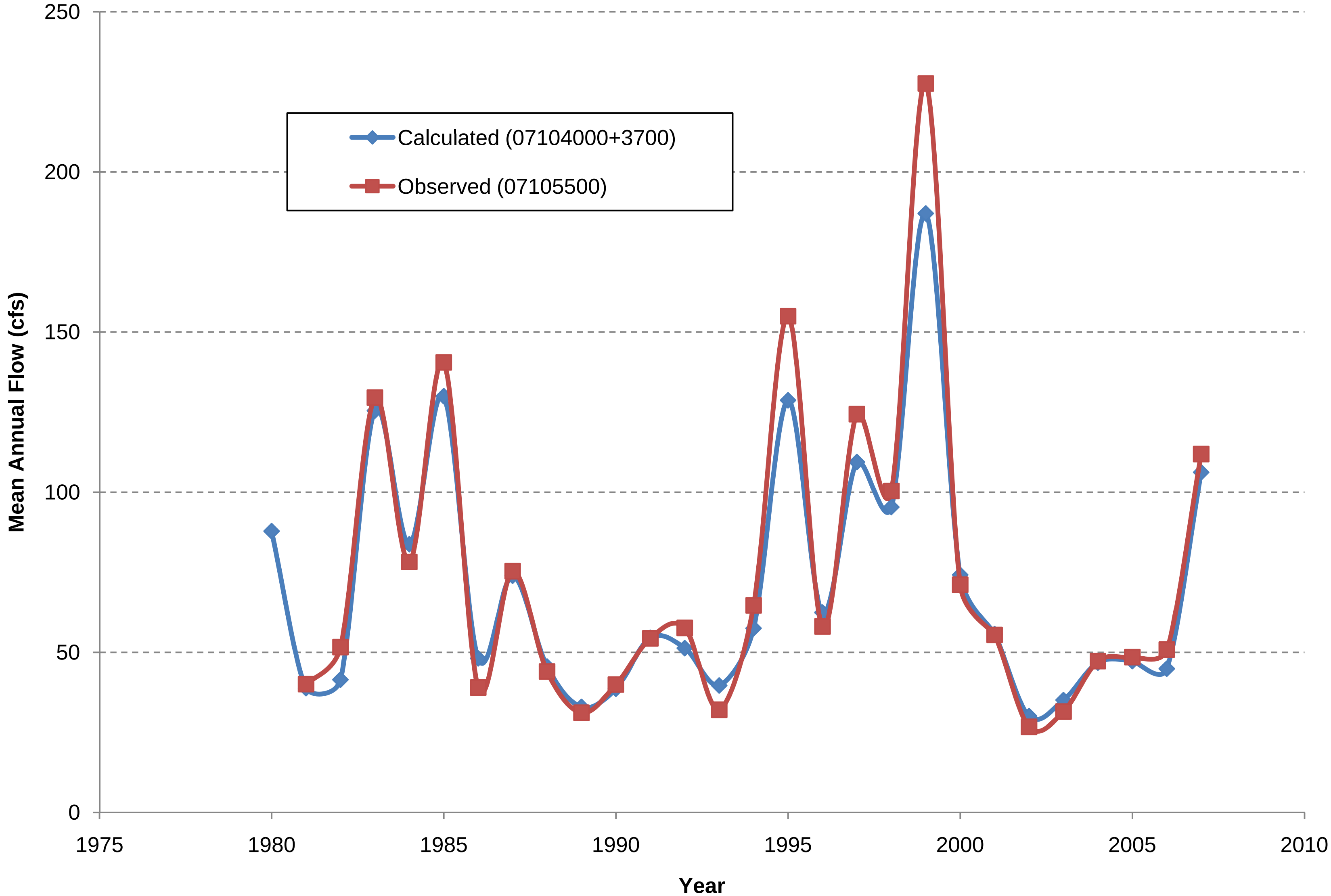


Figure 2-3: Mean Annual Flow at USGS Gage 07105500: Calculated vs. Observed

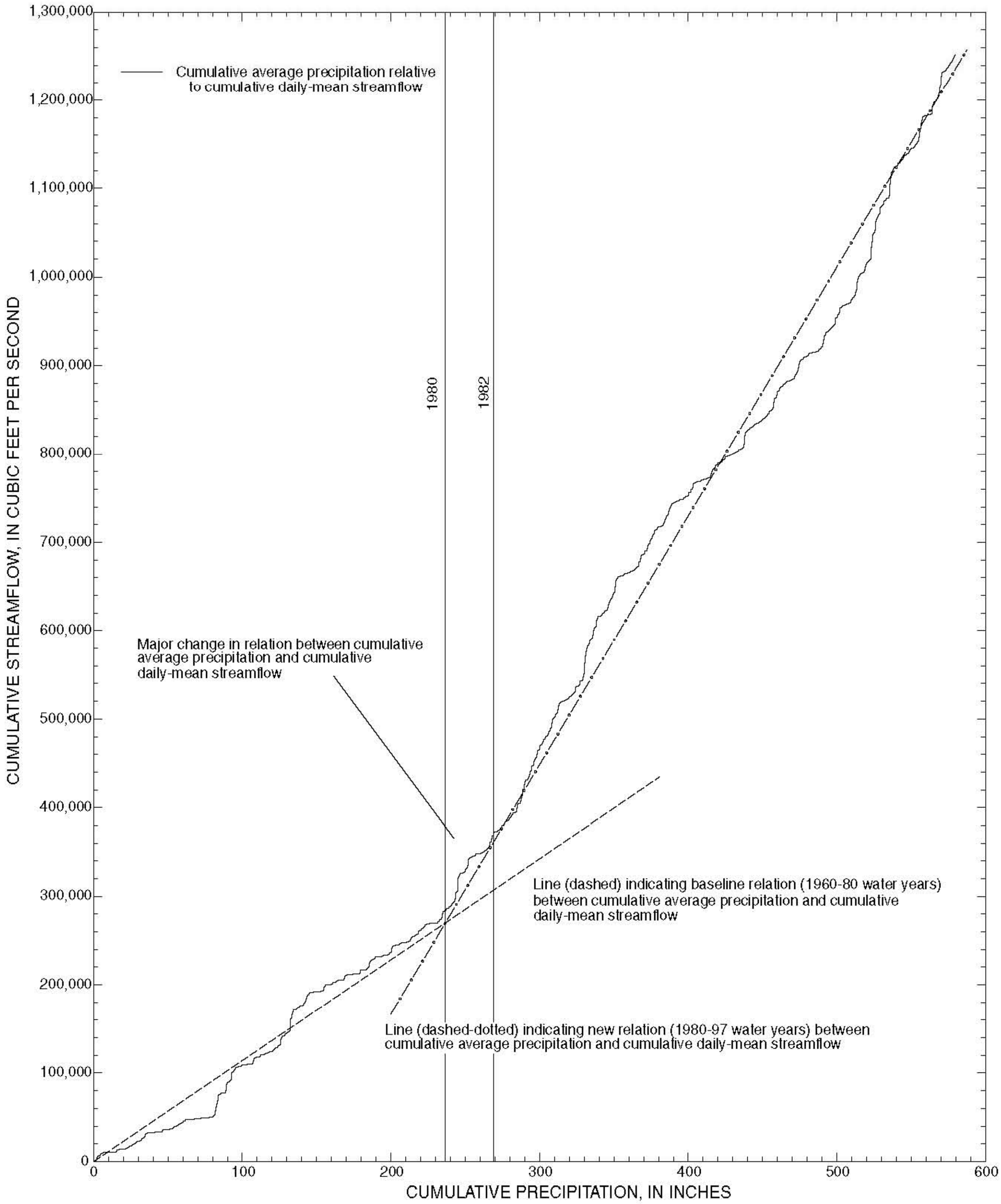


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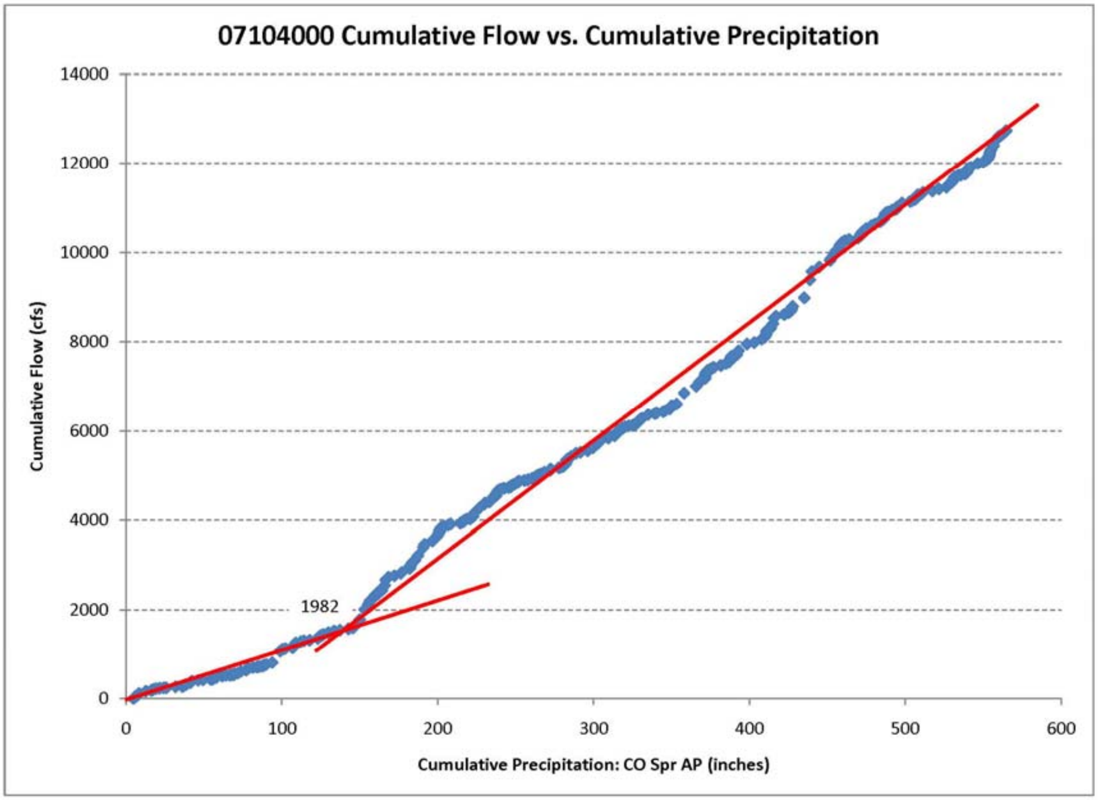
Figure 2-4. Double Mass Curve of Streamflow vs. Precipitation USGS Gage 07106500 Fountain Creek at Pueblo (USGS 2000)

CDM



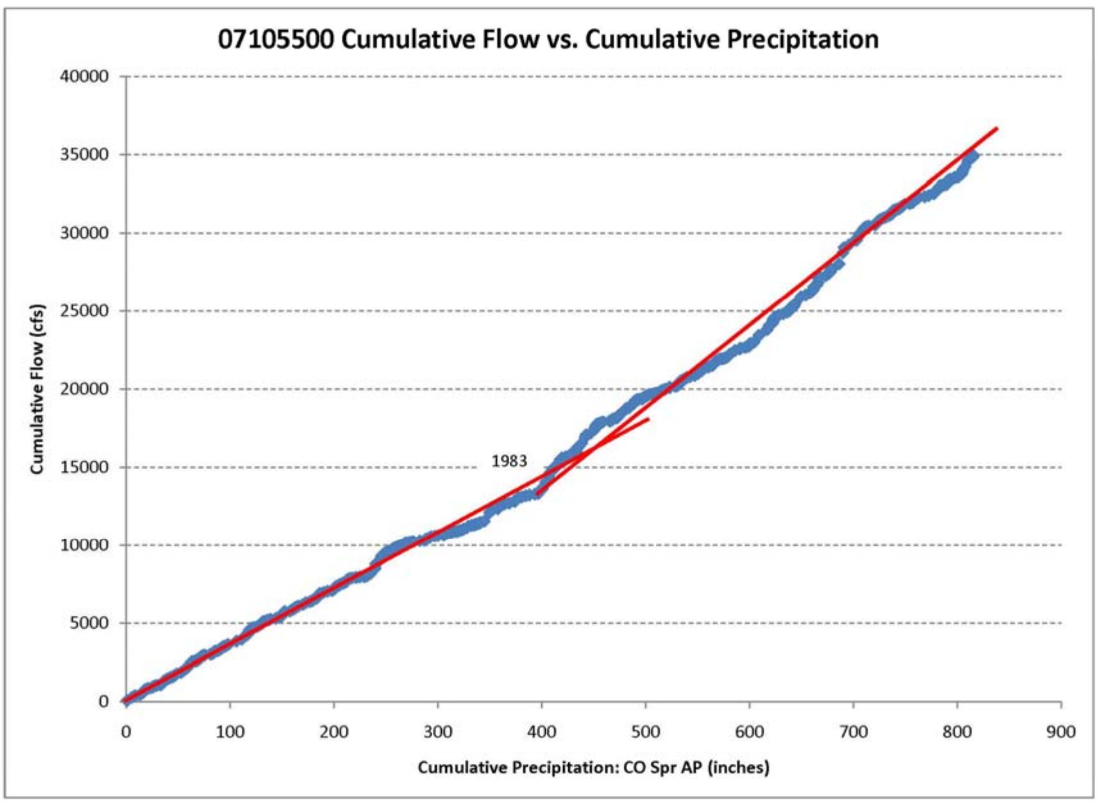


Figure 2-5: Double Mass Curve of Streamflow vs. Precipitation USGS Gages 07104000 (Monument Creek at Pikeview) and 07105500 (Fountain Creek at Colorado Springs)

DRAFT

CDM

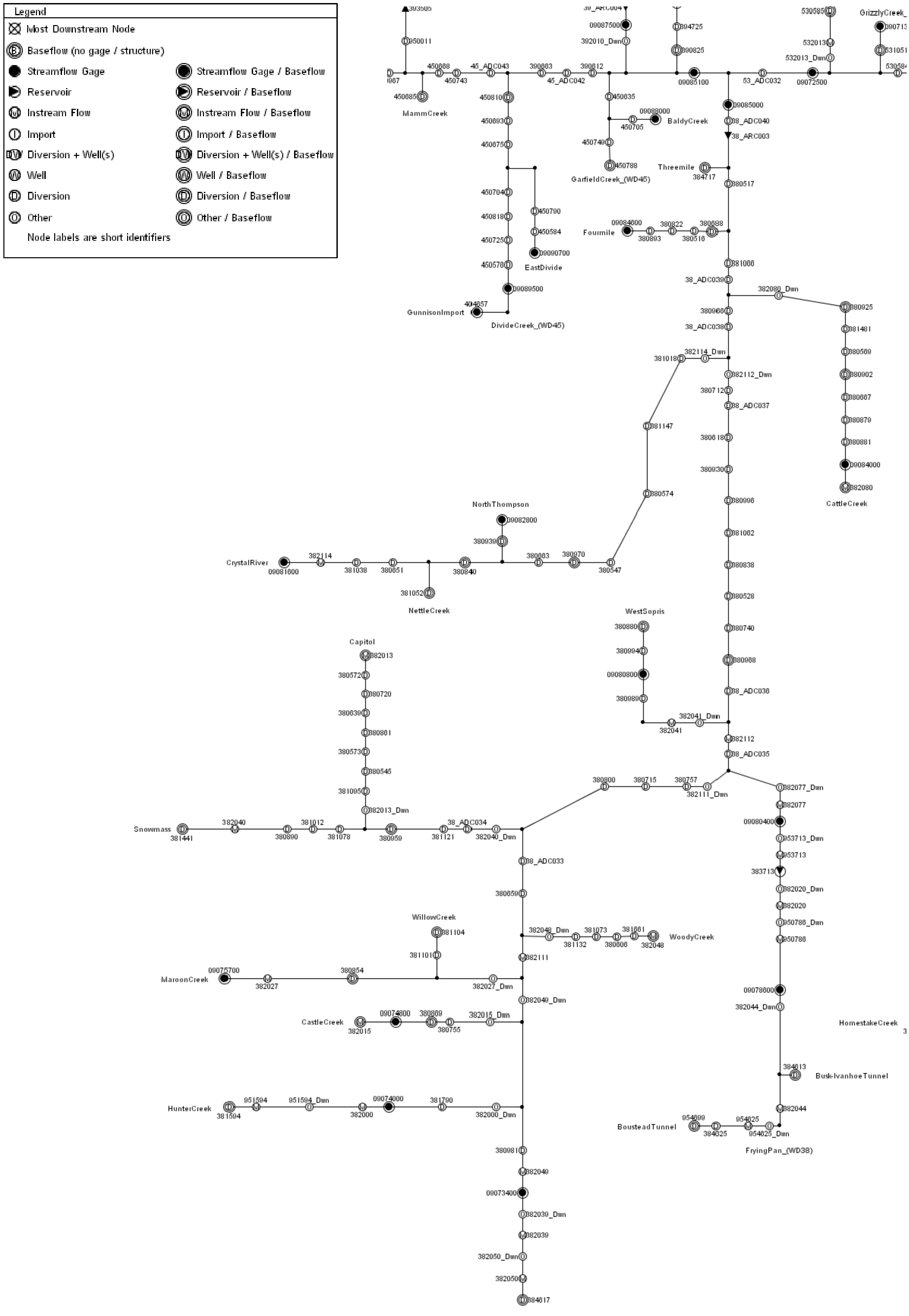


Figure 2-6 StateMod Network Diagram For Roaring Fork Watershed

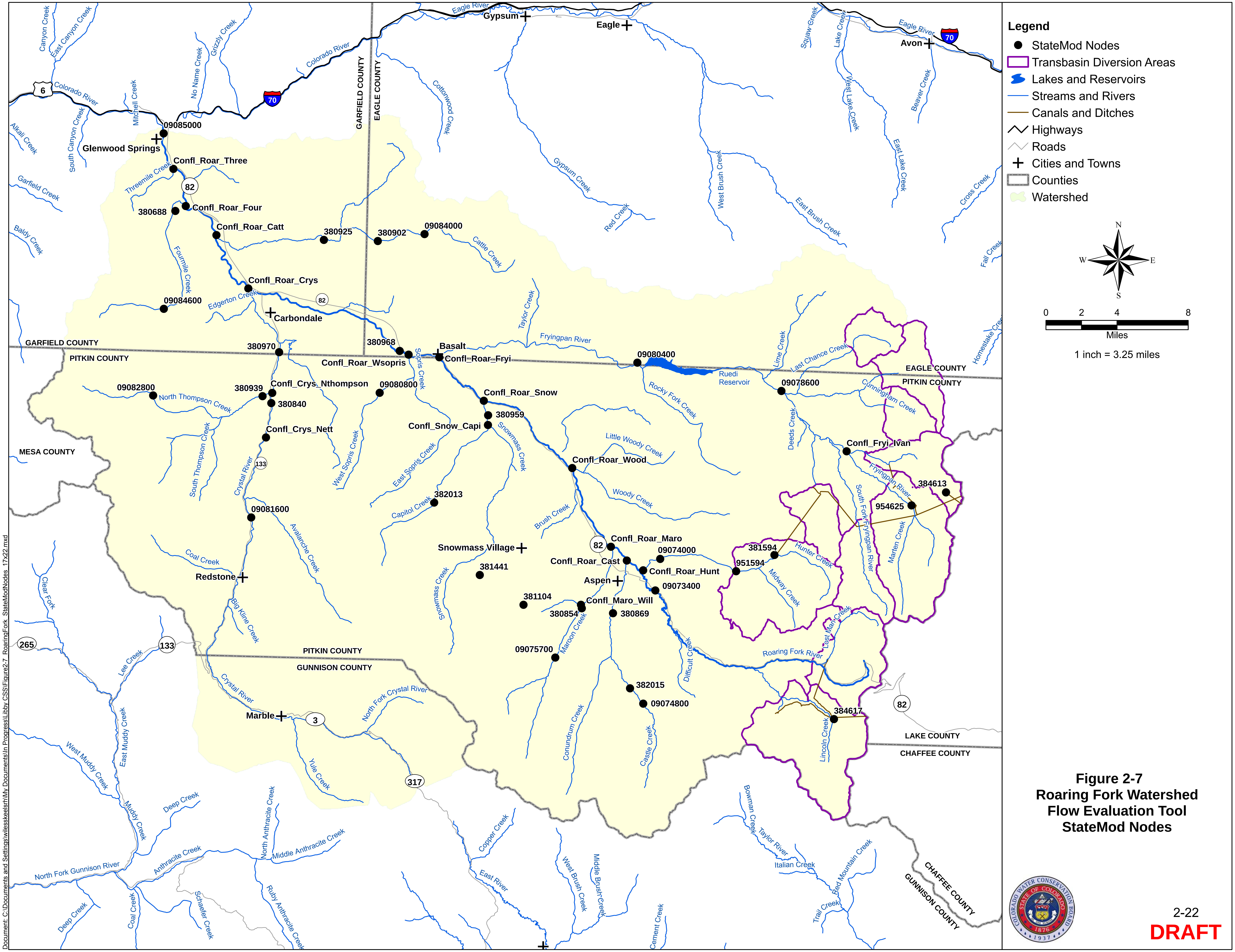


Figure 2-7
Roaring Fork Watershed
Flow Evaluation Tool
StateMod Nodes

Section 3
Watershed Flow Evaluation Tool Results

3.1 Fountain Creek Watershed Flow Evaluation Tool Results

The results of the WFET flow metric calculations and mapping and the ecological risk mapping for the Fountain Creek are discussed in this section. The flow metric calculations and associated mapping address the question "how have flows changed from baseline to existing conditions?" and the ecological risk mapping addresses the question "how do these flow changes relate to ecological changes or risk?"

3.1.1 Flow Metric Results

Figures 3-1 through 3-4 at the end of this section show the results of the mapping for the following flow metrics:

- Mean annual flow (Figure 3-1)
- Mean January flow (Figure 3-2)
- Mean August flow (Figure 3-3)
- One-day peak flow (Figure 3-4)

For the mean annual flow metric (Figure 3-1), flows increased from the baseline to existing conditions for all five USGS gage locations used in the analysis. The downstream gages showed higher increases than upstream gages. For the upper gages (0714000 and 0715500), the percent increases from baseline to existing conditions were between 30 and 45 percent. For the downstream gages (07105800, 07106000, and 07106500) increases were between 180 and 200 percent.

For mean January flow (Figure 3-2), flows also increased from baseline to existing conditions for all of USGS locations. Similar to the mean annual flow, downstream gages had higher increases than the upstream gages. For the upstream gages, the increases ranged from 50 to 80 percent. For the downstream gages, the increases were between 190 and 330 percent.

The mean August flow metric map (Figure 3-3) and the one-day peak flow metric map (Figure 3-4) show increases from baseline to existing conditions, but the percent of change varies throughout the watershed for both metrics. For mean August flow, percent increases ranged from 40 to 200 percent and for the one-day peak flow, percent increases were between 20 and 100 percent.

3.1.2 Ecological Risk Mapping Results

Figures 3-5 through 3-12 and 3-15 and 3-16 at the end of this section display the results of the ecological risk mapping for the following ecological attributes in the Fountain Creek Watershed:

- Trout (Figures 3-5 through 3-8)
- Warm water fishes (Figures 3-9 through 3-12)
- Erosion potential (Figures 3-15 through 3-16)

For trout, only the uppermost gages were considered in the ecological risk mapping as this portion of the Fountain Creek watershed is considered in the transition zone from a mountain to plains stream where trout can still be expected. Figure 3-5 exhibits the baseline ecological risk for trout. Both USGS gages mapped for trout show that the ecological risk for trout at the two locations is minimal based on flow conditions. Figure 3-6, the existing ecological risk map for trout, also shows that the ecological risk is low. It should be noted that the existing condition metric calculations are significantly greater than 55 percent, which is the upper limit of the risk level presented in Figure 3-6. The data and literature documented in Appendix B for trout address ecological risk associated with greatly increased summer low flows for trout. The effect of late-summer flow augmentation on trout as well as native fishes may merit additional research. Figure 3-7 and 3-8 show the inferences between USGS gages for the trout metric as outlined in the Section 2.4.7.

For warm water fishes, all gages in the watershed were considered for the ecological risk mapping. Figure 3-9 displays the baseline ecological risk for warm water fishes, and Figure 3-10 shows the existing ecological risk for warm water fishes. Both the baseline and existing conditions indicate minimal risk for warm water fishes. Similar to the trout results, the existing condition metric calculations are significantly greater than 40 percent or the upper limit of the risk level. Like the trout results, the literature search results (Appendix B) do not provide insight into the ecological risk associated with significant increases in summer low flows. Figures 3-11 and 3-12 show the potential ecological risk levels between USGS gages.

To calculate the erosion potential in Fountain Creek, sediment transport capacity of the channel was estimated. The sediment transport capacity of flows less 2300 cfs at the Fountain Creek at Fountain gage has increased approximately fivefold in the post-1980 period compared to the pre-1970 period (Figure 3-13). Bankfull discharge has been previously estimated at 3100 cfs (URS 2007), but the most erosive flows over time now appear to be in the range of 100 – 1000 cfs. Results from the streamflow records at the Pueblo gage (Figure 3-14) also suggest a four- to fivefold increase in the cumulative sediment transport capacity of sub-bankfull flows (<3000 cfs). In Figures 3-13 and 3-14, the areas under the curves representing the pre-1970 and post-1980 periods provide a relative comparison of the cumulative sediment transport capacity for the two time periods. The highest point in an effectiveness curve represents an estimate of the effective discharge, the flow that transports the largest portion of the annual sediment yield over a period of years (Andrews, 1980). The absolute values on the vertical axes differ between plots because channel bed material and the appropriate sediment transport relationships used for the Magnitude-Frequency Analysis differed between gage locations. As such, the two curves are best interpreted in terms of differences in relative sediment transport capacity between the two periods as opposed to absolute values across sites.

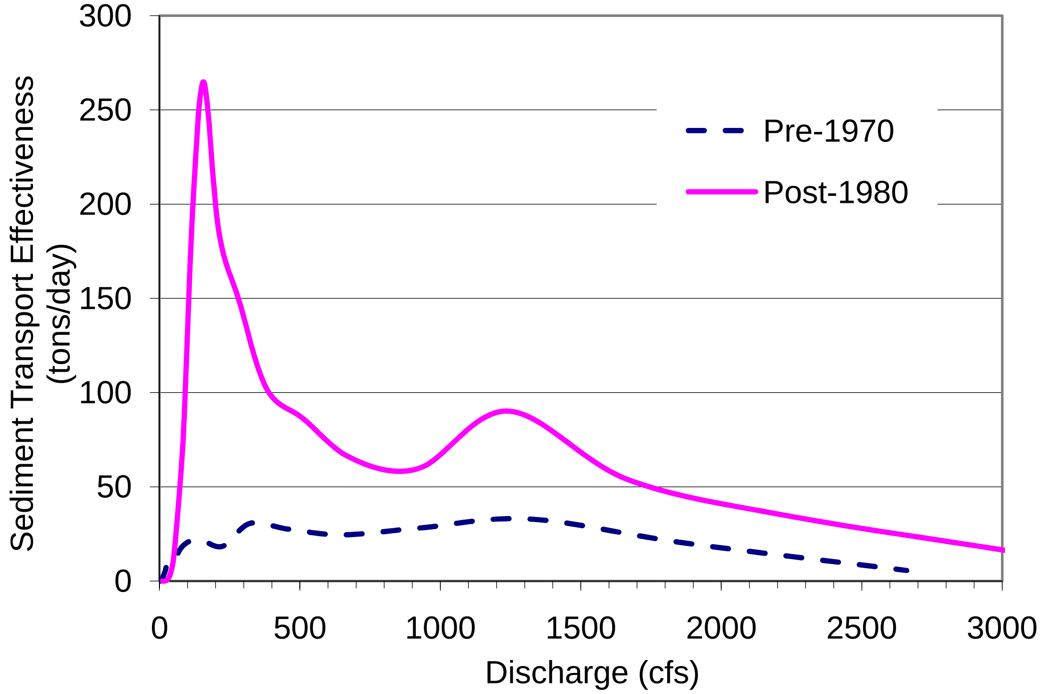


Figure 3-13. Sediment Transport Effectiveness Curves for Fountain Creek at Fountain, Pre-1970 vs. Post-1980

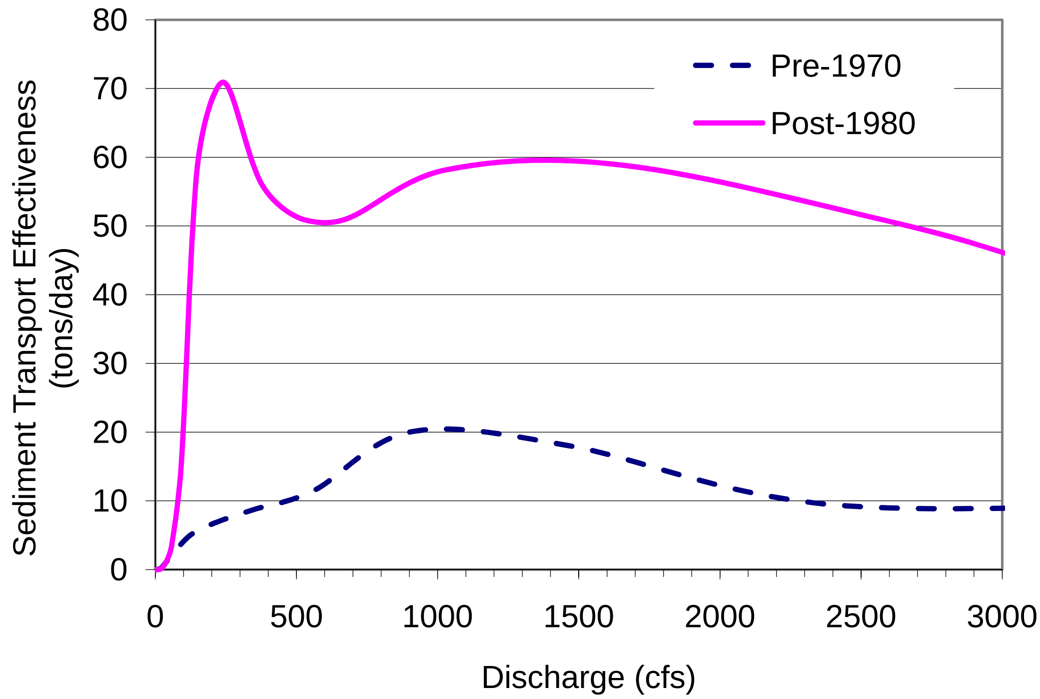


Figure 3-14. Sediment Transport Effectiveness Curves for Fountain Creek at Pueblo, Pre-1970 vs. Post-1980

An effective discharge analysis was conducted as part of a previous study of Fountain Creek (URS 2007). As in the present study, the MFA performed by URS (2007) indicates that calculated effective discharges are generally much smaller than field estimated bankfull discharges. As opposed to focusing on the ranges of flows associated with most of the long-term sediment transport in the mainstem of Fountain Creek, the previous study shifted its focus to bankfull flows (URS, 2007, p. 2 – 56).

> Because the calculated effective discharges are uncharacteristically low, the bankfull discharges were then selected to calculate the representative sediment load for aggradation/degradation tendency evaluation.

In addition to the previous focus on bankfull geomorphic characteristics, the results of this preliminary analysis underscore the importance of accounting for the influence of moderate, sub-bankfull flows when considering management options for lower Fountain Creek, whether for ecological conservation or mitigating channel instability. Ultimately, the high spatial and temporal variability of channel forms along the Fountain Creek mainstem precludes assessments of riparian maintenance flows and other flow-ecology relationships without detailed site-specific information on channel morphology and floodplain characteristics. Such an analysis is entirely feasible but is beyond the scope of this pilot study focused at coarser scales.

Figures 3-15 and 3-16 show the erosion potential risk maps for Fountain Creek. The erosion potential mapping categories were estimated by calculating the ratio of the existing erosion potential to baseline erosion potential. Figures 3-15 and 3-16 show that the erosion potential of sub-bankfull flows has been magnified approximately four- to fivefold downstream of Colorado Springs based on the above analysis.

3.2 Roaring Fork Watershed Flow Evaluation Tool Results

The results of the WFET flow metric calculations and mapping and the ecological risk mapping for Roaring Fork are discussed in this section. Similar to the Fountain Creek Watershed above, the flow metrics calculations and associated mapping address the question "how have flows changed from baseline to existing conditions?" and the ecological risk mapping addresses the question "how do these flow changes relate to ecological changes or risk?" Second, results from the site-specific quantification for the Roaring Fork River between Basalt and Carbondale that are detailed in Appendix A are compared to the WFET results.

3.2.1 Flow Metric Results

Figures 3-17 through 3-20 at the end of this section show the results of the mapping for the following flow metrics:

- Mean annual flow (Figure 3-17)
- Mean January flow (Figure 3-18)

- Mean August flow (Figure 3-19)
- One-day peak flow (Figure 3-20)

For the mean annual flow metric (Figure 3-17), flows decreased from the baseline to existing conditions for all node locations used in the watershed, although for 12 of the 47 nodes the decrease was less than 10 percent which may represent effectively no change. Percent decreases were higher in the headwaters with the highest percent decreases occurring in Cattle Creek. The southern portion of the watershed showed the lowest percent decreases. The mean January flow metric (Figure 3-18) indicates that January flows have both decreased and increased from baseline to existing conditions throughout the watershed. The percent decreases occur in headwaters (below major diversions), and the percent increases occur in the lower portion of the watershed (due to reservoir releases or agricultural return flows). The highest percent increases are found downstream of Ruedi Reservoir. For the mean August flow metric (Figure 3-19), most areas in the watershed show decrease from baseflow flow to existing conditions. Mean August flow has increased downstream of Ruedi Reservoir. Throughout the watershed, one-day peak flows (Figure 3-20) have decreased from existing to baseline conditions, although for 21 of the 47 nodes the decrease is less than 10 percent, which may be insignificant. The majority of the percent decreases are slight with some higher decreases occurring in the headwaters.

3.2.2 Ecological Risk Mapping Results

Figures 3-21 through 3-32 at the end of this section display the results of the ecological risk mapping for the following ecological attributes in the Roaring Fork watershed:

- Trout (Figures 3-21 through 3-24)
- Flannelmouth sucker (Figures 3-25 through 3-26)
- Recreation (Figures 3-27 through 3-30)
- Riparian (Figures 3-31 through 3-33)

For the trout risk mapping, Figure 3-21 shows the ecological risk for baseline conditions, and Figure 3-22 shows the ecological risk for existing conditions. Figures 3-23 and 3-24 show the estimated ecological risk between nodes. For both baseline and existing conditions, the flow metrics indicated that throughout much of the watershed ecological risk for trout is low. However, the risk for trout has increased to moderate levels for the Roaring Fork between Hunter and Castle creeks, as well as in Cattle Creek. The results also indicate that Lincoln Creek has also increased in risk from the low to the minimal risk category.

Flannelmouth sucker are found in the lower portion of the Roaring Fork watershed (Figures 3-25 and 3-26). The ecological risk for Flannelmouth sucker is low through the stretch of river where they occur.

Figures 3-27 and 3-28 show the recreation risk mapping for the minimum flow metric described in Section 2.4.5. These figures show for baseline and existing conditions the number of months between May and August that on average exceed the minimum flow metric detailed in Section 2.4.5. The two figures show that baseline conditions have less risk in headwater areas than existing conditions. Figures 3-29 and 3-30 display risk mapping for the preferred flow metric explained in Section 2.4.5. Similar to the minimum flow metric risk results, the upper portions of the watershed have less risk in the baseline conditions than existing conditions.

Figures 3-31 and 3-32 show the results of the riparian ecological risk mapping. These maps show that risks to riparian areas is highest in the upper portions of the watershed and downstream of Ruedi Reservoir. Ecological risk for riparian areas is lowest in the southern portion of the watershed.

3.2.3 Validation with Site-Specific Results

An initial validation of the WFET with site-specific data was completed for one location at the Roaring Fork River between Basalt and Carbondale. The WFET results show that for baseline and existing conditions there is minimal ecological risk for trout in this segment (Figures 3-21 through 3-24). The site-specific results described in detail in Appendix A show similar results. For the site-specific analysis, there is little difference in wetted area or water depth between the existing conditions and baseline conditions in August and September, regardless of channel type (Figures 3-33 through 3-36). Both riffle and run channel types are wet from bank to bank at the baseline and existing flows. The fully wetted channel provides good conditions for fish and invertebrates. Since the habitat conditions are similar between baseline and existing flows, the trout habitat should be similar. This river reach has very good trout populations under existing conditions.

Another part of the validation effort included examining conditions that would be considered as high ecological risk for trout. The WFET was used to determine flows that should be a high risk for trout populations. These flows were 48 cfs, 98 cfs, and 148 cfs, which correspond to poor, marginal, and fair habitats based on the WFET metrics. An evaluation of those flows was made using the site-specific data. Total wetted area as a function of discharge shows that wetted area at all three flows is substantially lower in comparison to wetted area at the existing August-September average flow of 730 cfs (Figure 3-37). The decrease in wetted area at lower flows is the result of the channel shape and the stage discharge function. There is a very large change in water surface with small changes in flows up to the point where the majority of the channel is wet (Figure 3-38). This relationship is also shown with the comparison of wetted area. There is a substantial difference between the percent of the channel that is wet at the high-risk flow and the percent wet at the existing flow (Figure 3-39). The comparison of the water surface elevations for the high-risk and existing flows shows that this relationship holds for most of the individual cross sections (Figures 3-40 through 3-44).

Figures 3-39 through 3-44 also indicate that site-specific information can be used to further refine the WFET ecological risk levels. These figures show that there is greater wetted area loss when flows fall below 300 cfs. The WFET ecological minimal risk level (yellow bar in Figures 3-40 through 3-44) may be too low at this location as the moderate and minimal ecological risk levels show similar reduction in wetted area loss.

The habitat versus discharge relationship does not show the same stage versus discharge relationship. There is approximately the same total amount of habitat for flows less than 100 cfs and 730 cfs (Figure 3-45). The maximum habitat for adult trout occurs at approximately 300 cfs. While the habitat quantity may be similar at the lower and higher flows, the habitat quality is different. The Physical Habitat Simulation System (PHABSIM) calculates the weighted usable area by multiplying the area represented by a particular location by the combined suitability for depth, velocity, and substrate. This results in habitat quantification for the site that varies by location. In the comparison of 48 cfs with 730 cfs, the habitat at 48 cfs is lower quality, and less river channel is available than at 730 cfs (Figure 3-46). There is approximately 10 percent difference in total usable habitat for rainbow trout adults between 48 cfs and 730 cfs. However, the habitat at 730 cfs is of higher quality (see lower legend Figures 3-46 and 3-47 for habitat quality). The legend shows the habitat quality by cross-section. The habitat quality can be depicted in a three-dimensional view as well to display channel shape in combination with habitat quality (Figures 3-48 and 3-49). The above examples show that the site-specific approach can provide a robust characterization of habitat between two different flows, and an appropriate approach should compare total wetted channel and change in water surface, habitat quantity, and habitat quality. The PHABSIM model has built-in graphics for all of these metrics; however, the graphics are preset for the displays, axis titles, and legends. The units for all the PHABSIM data are English (feet or square feet).

The most appropriate interpretation of the PHABSIM data should include a comparison of the multiple metrics that can be derived from the PHABSIM model including wetted area, wetted perimeter, and weighted usable area (quality and quantity) at multiple flows. These multiple hydraulic and habitat metrics should be included in the validation and calibration of future WFET applications.

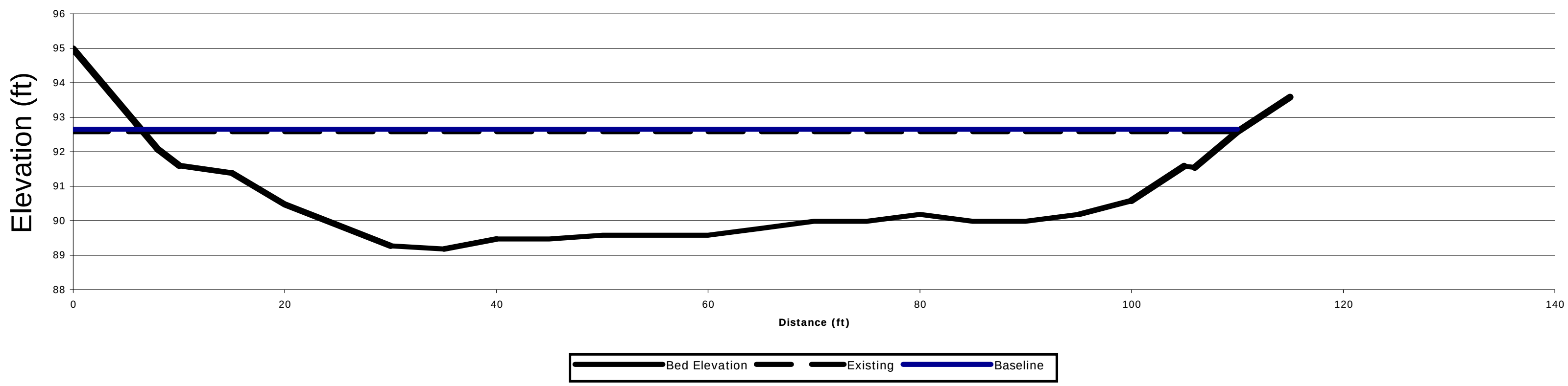


Figure 3-33. Comparison of Baseline and Existing Water Surface Elevation for Roaring Fork River Cross Section 2 for Average August Discharge

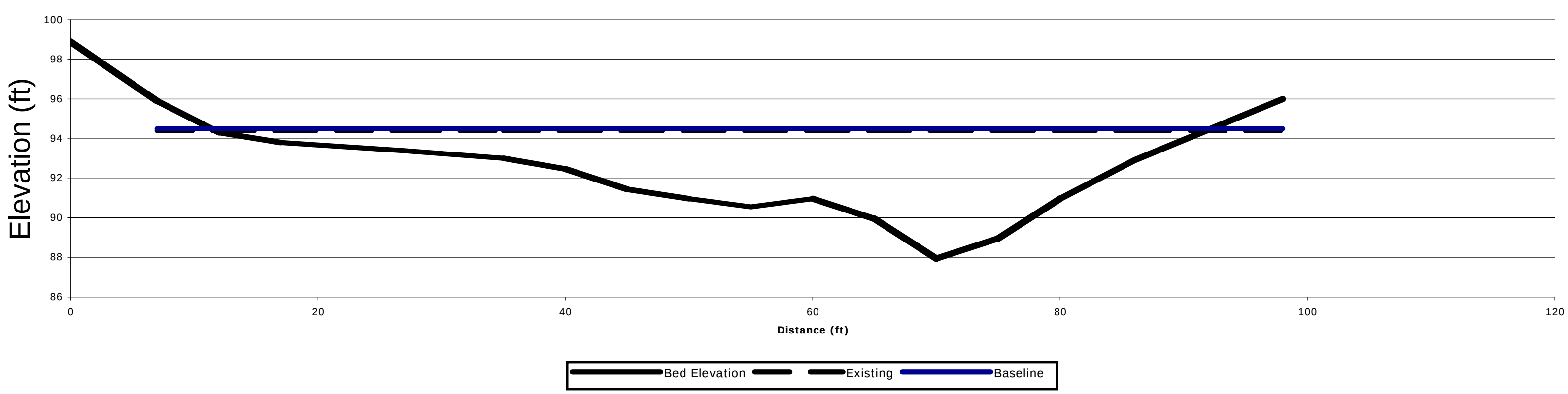


Figure 3-34. Comparison of Baseline and Existing Water Surface Elevation for Roaring Fork River Cross Section 4 for Average August Discharge

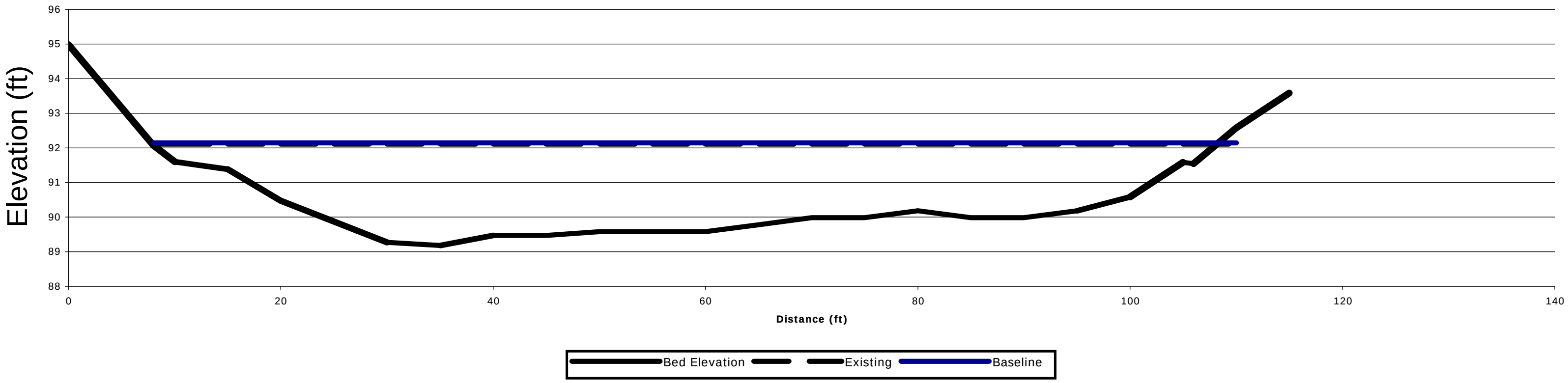


Figure 3-35. Comparison of Baseline and Existing Water Surface Elevation for Roaring Fork River Cross Section 2 for Average September Discharge

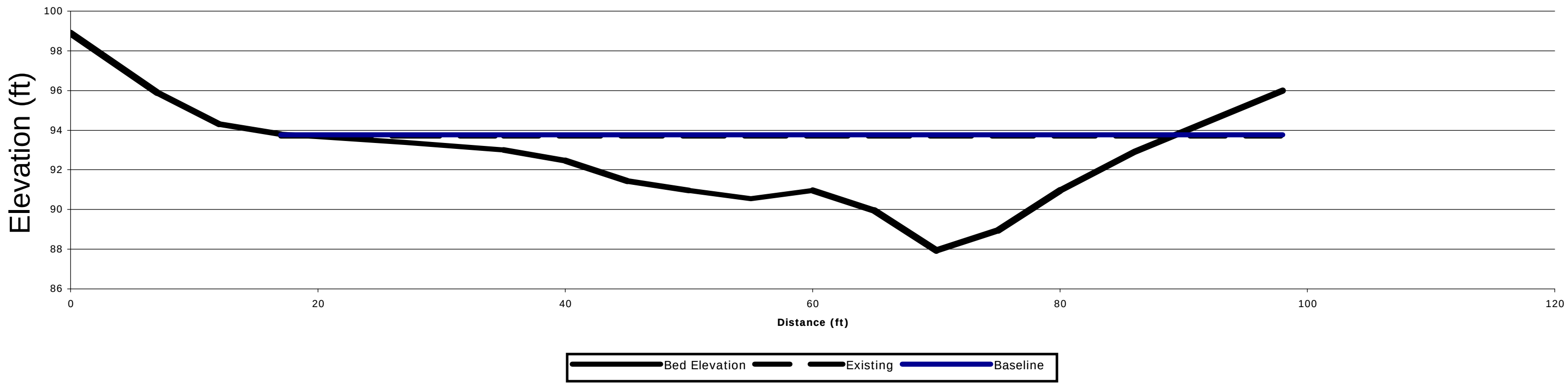


Figure 3-36. Comparison of Baseline and Existing Water Surface Elevation for Roaring Fork River Cross Section 4 for Average September Discharge

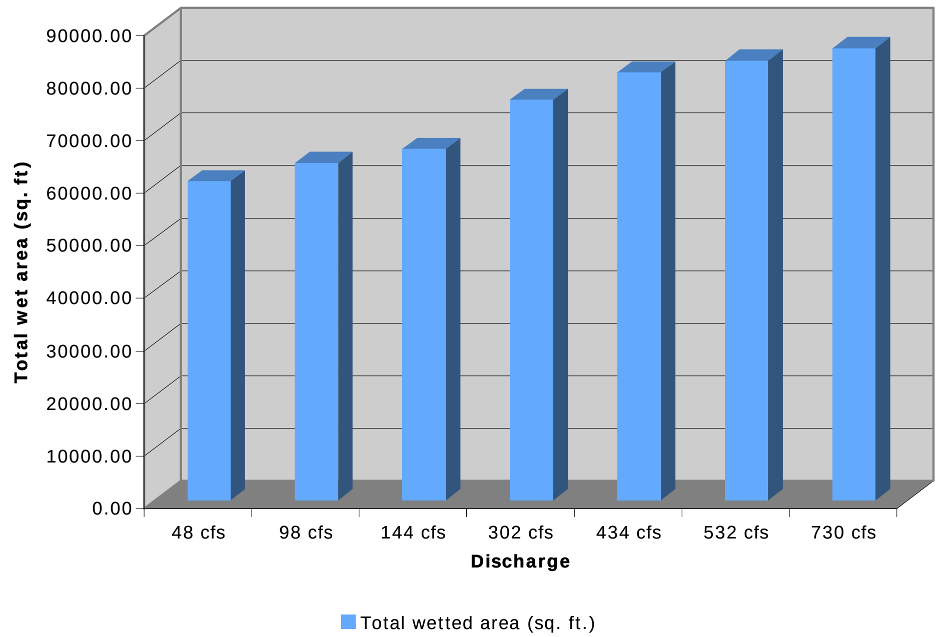


Figure 3-37. Total Wetted Area as a Function of Discharge for Roaring Fork River Site Roaring Fork River at Tree Farm (RFR-TF)

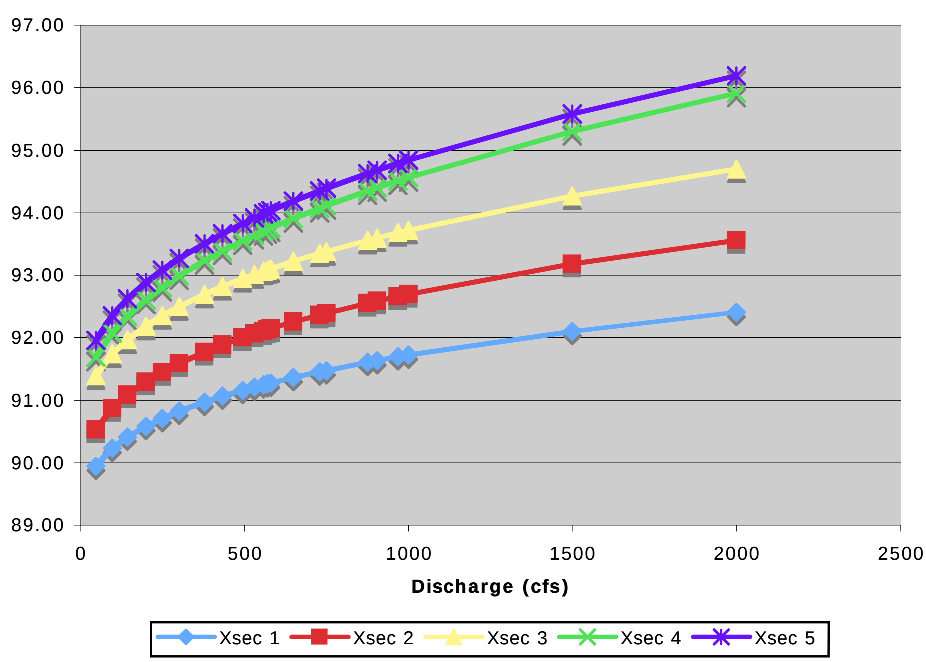


Figure 3-38. Stage Discharge Function for Site RFR-TF

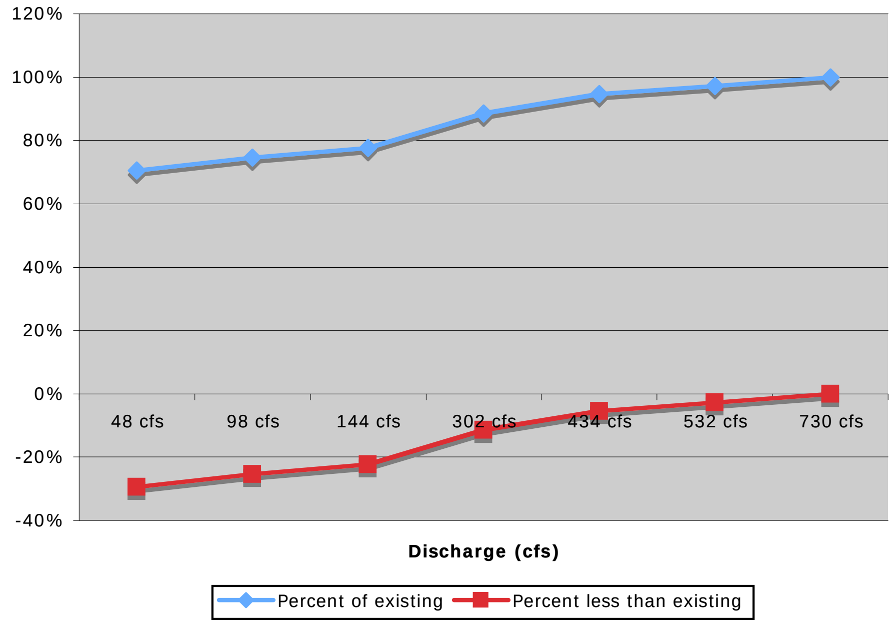


Figure 3-39. Percent Wetted River Channel at High-Risk Flows Compared to Existing Conditions at Site RFR-TF

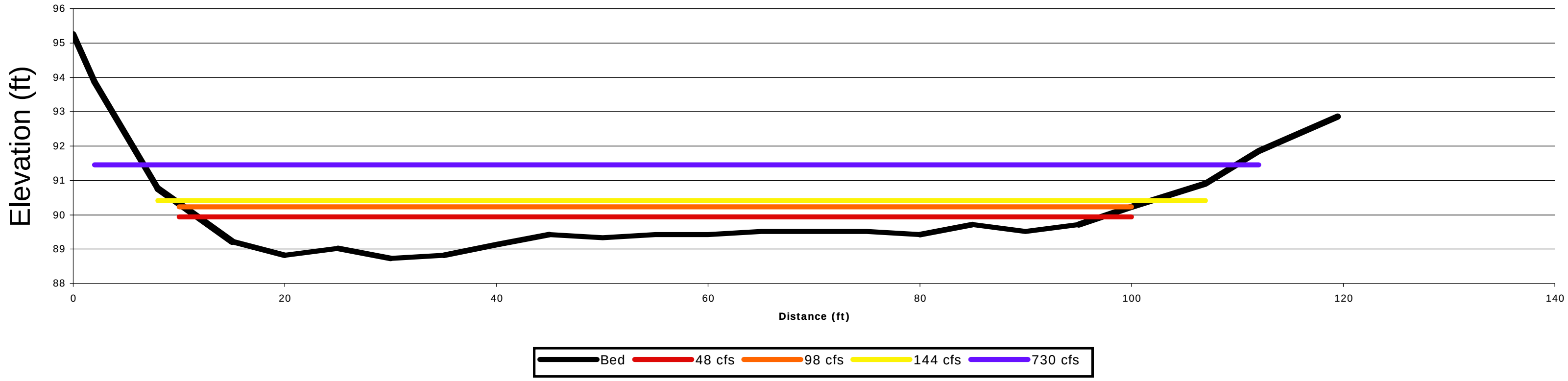


Figure 3-40. Comparison of Water Surface Elevations for High-Risk Flows and Average August-September Flows for RFR-TF Cross Section 1

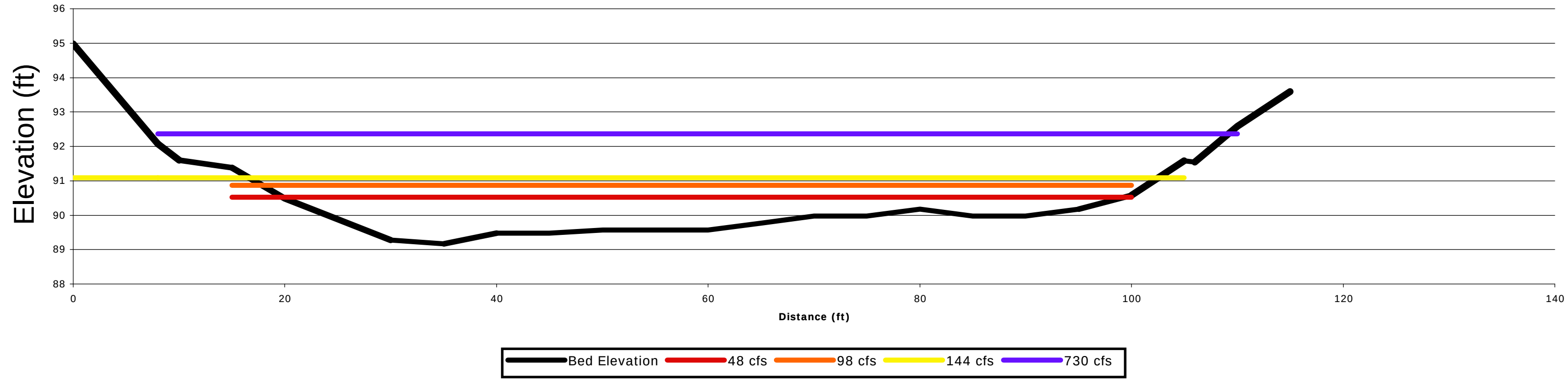


Figure 3-41. Comparison of Water Surface Elevations for High-Risk Flows and Average August-September Flows for RFR-TF Cross Section 2

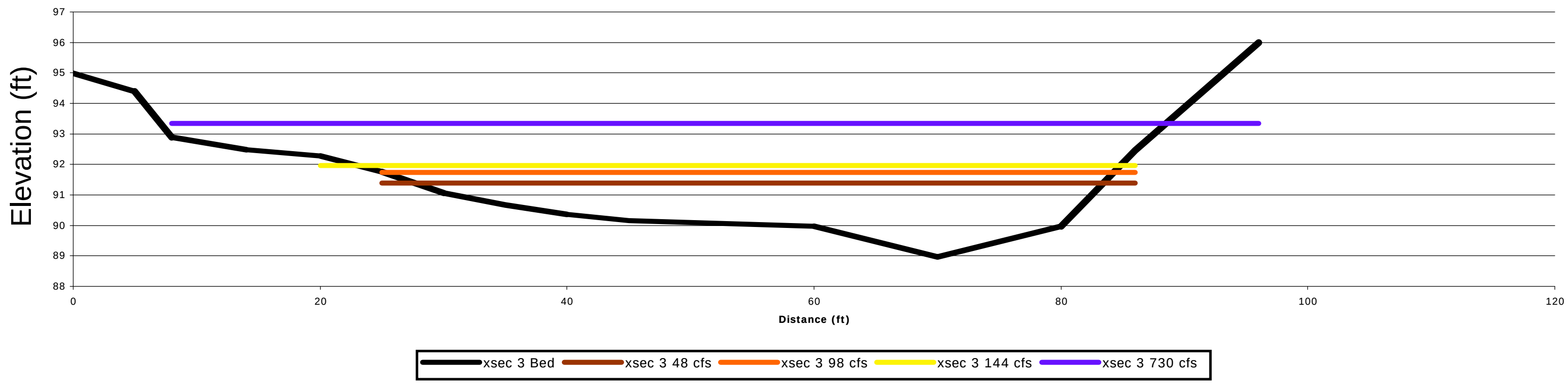


Figure 3-42. Comparison of Water Surface Elevations for High-Risk Flows and Average August-September Flows for RFR-TF Cross Section 3

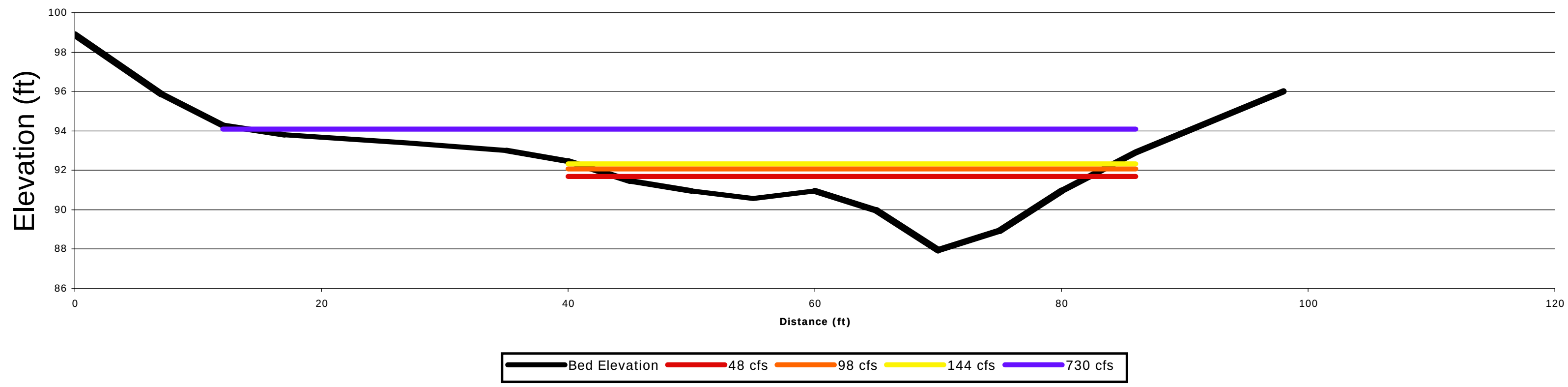


Figure 3-43. Comparison of Water Surface Elevations for High-Risk Flows and Average August-September Flows for RFR-TF Cross Section 4

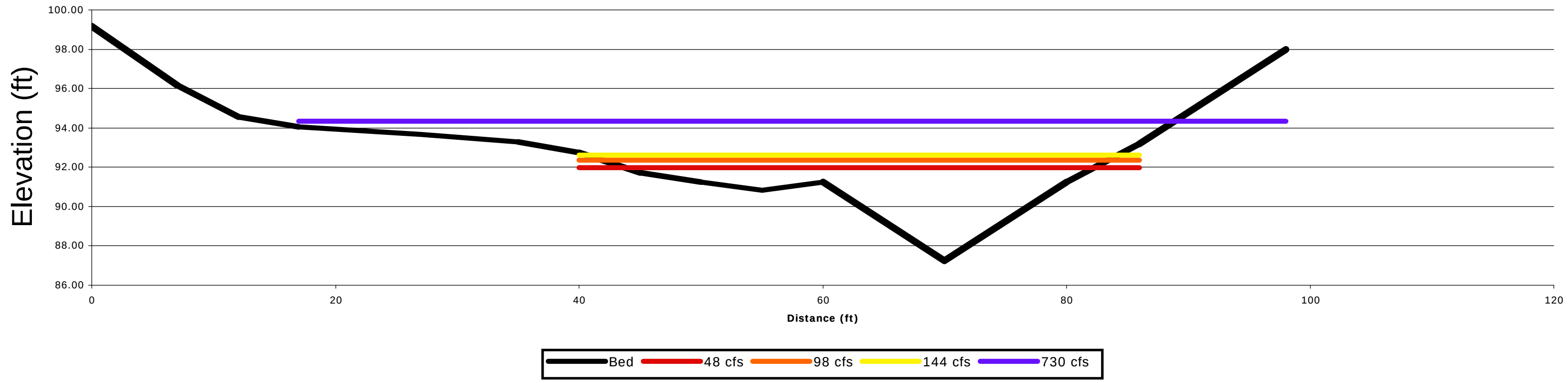


Figure 3-44. Comparison of Water Surface Elevations for High-Risk Flows and Average August-September Flows for RFR-TF Cross Section 5

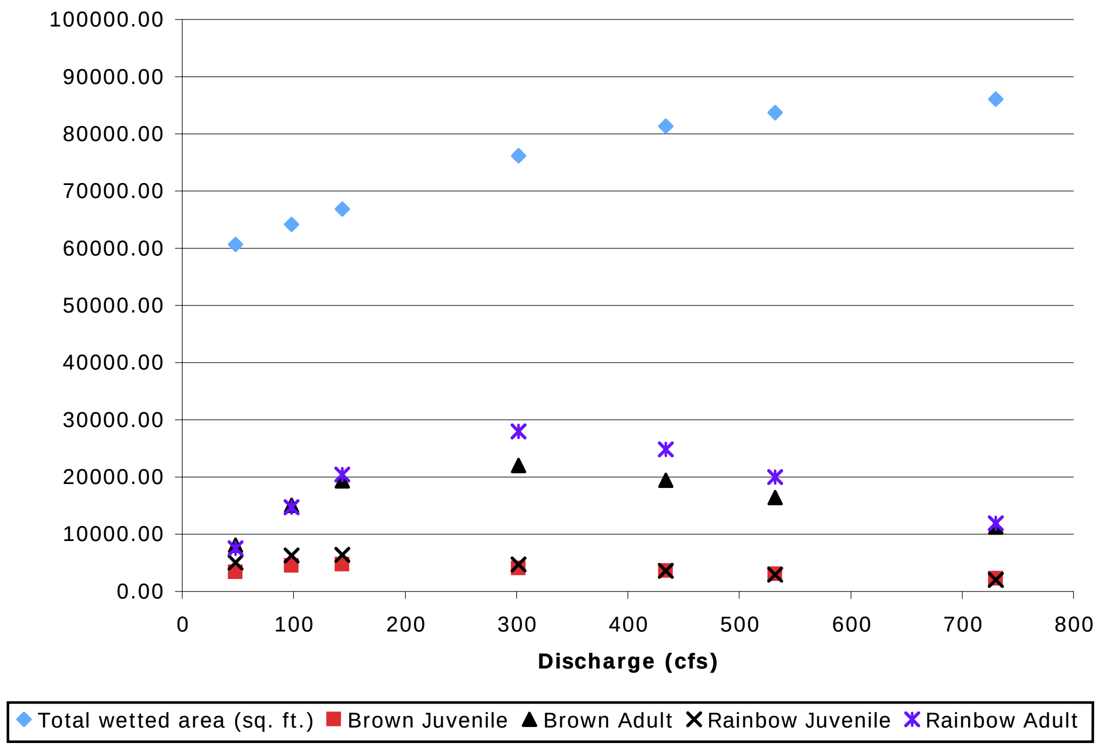


Figure 3-45. Total Wetted Area and Weighted Usable Area for High-Risk and Average August-September Flows at Site RFR-TF

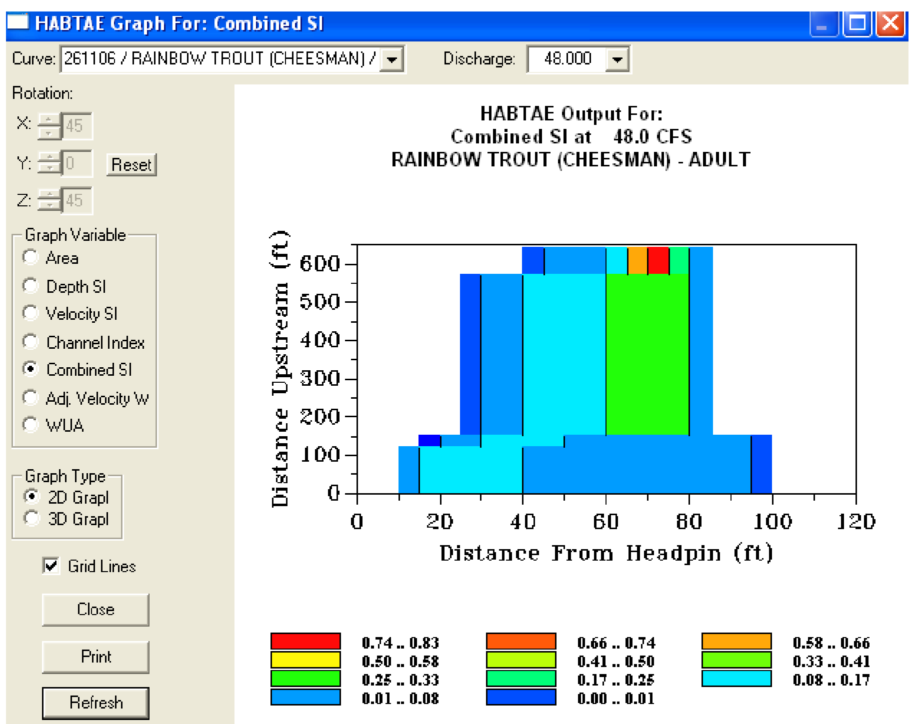


Figure 3-46. Example of Plan View of Rainbow Trout Habitat for Site RFR-TF at 48 cfs

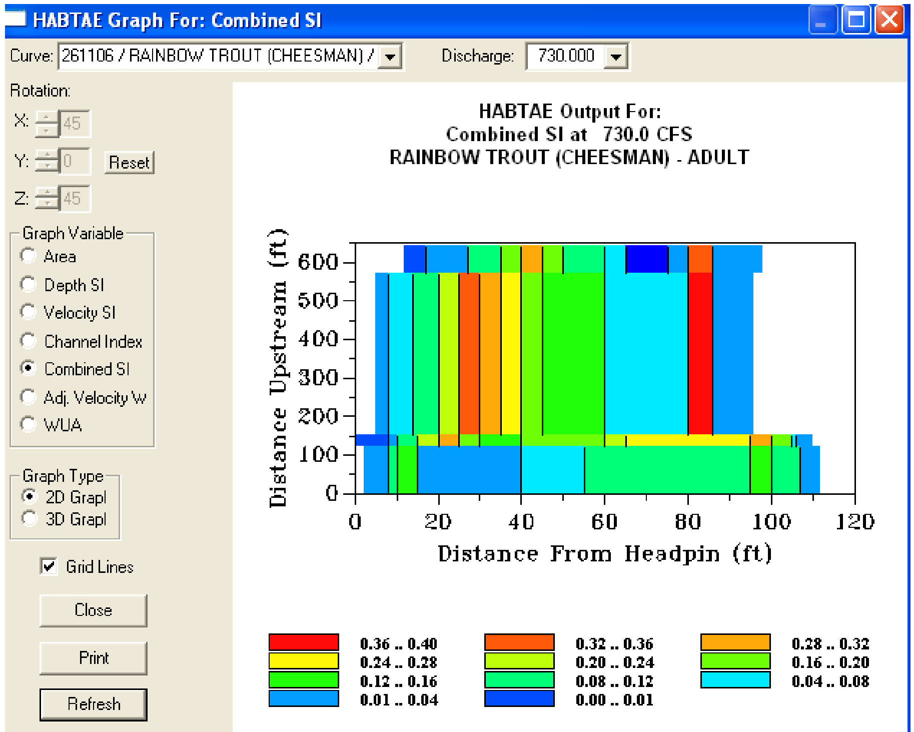


Figure 3-47. Example of Plan View of Rainbow Trout Habitat for Site RFR-TF at 730 cfs

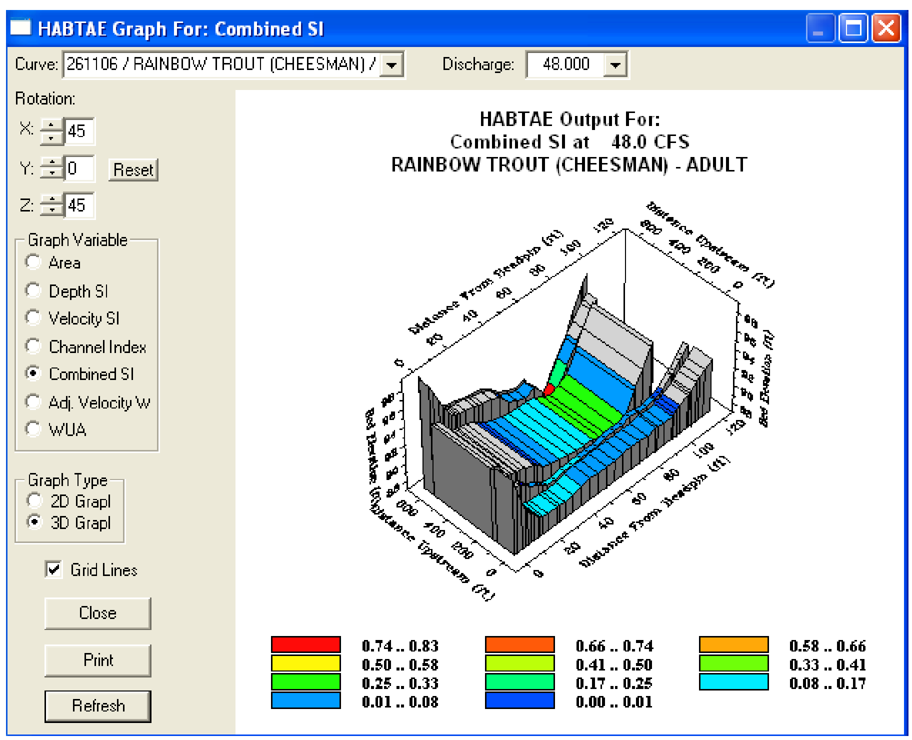


Figure 3-48. Example of 3-Dimensional View of Rainbow Trout Habitat for Site RFR-TF at 48 cfs

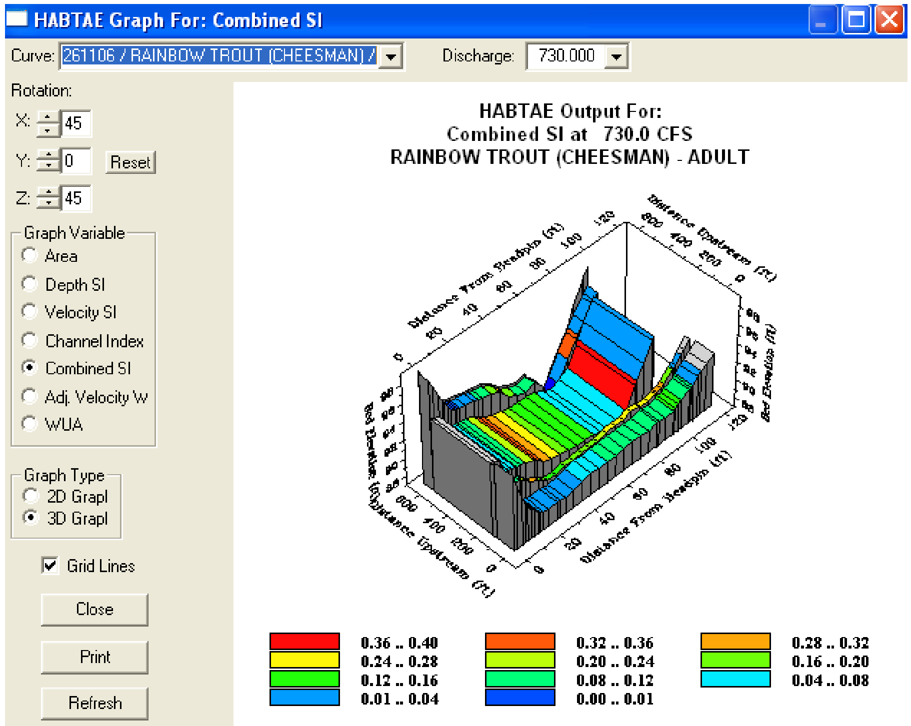


Figure 3-49. Example of 3 Dimensional View of Rainbow Trout Habitat for Site RFR-TF at 730 cfs

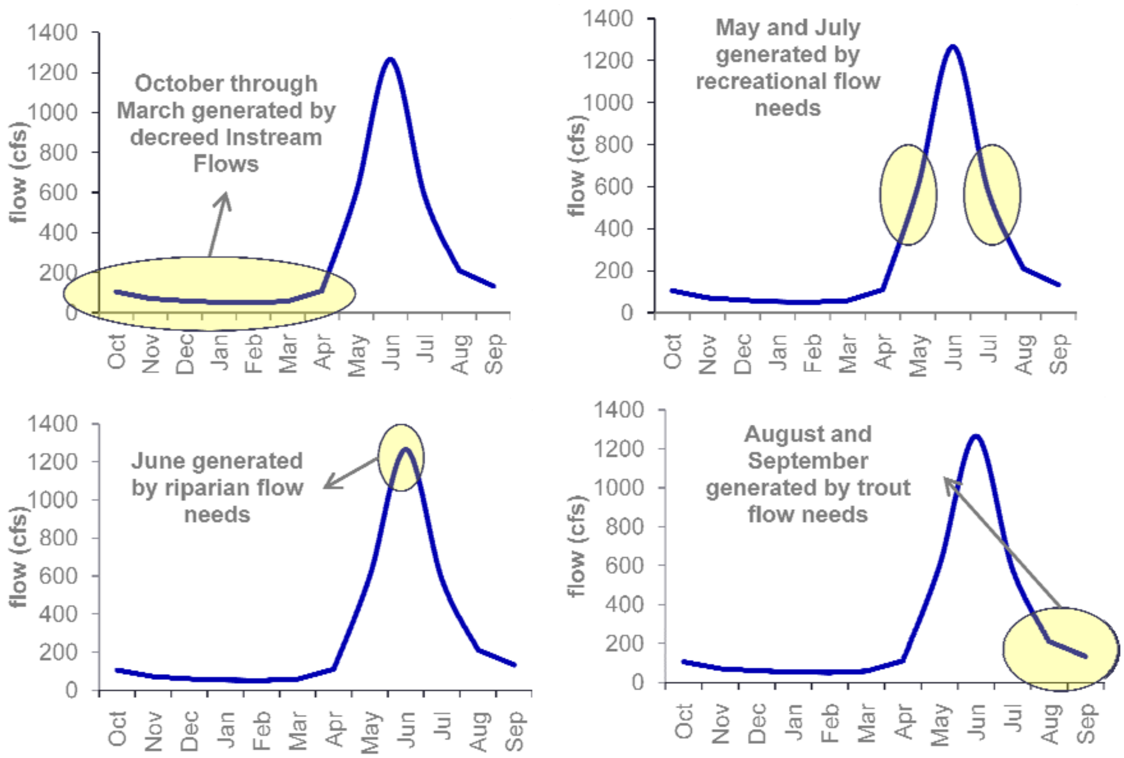
3.3 Application of the Watershed Flow Evaluation Tool for Developing Risk-Based Seasonal Flow Conditions

One potential application of the WFET is to estimate risk-based seasonal flow conditions that could be utilized in the Colorado River Water Availability Study or other water supply planning efforts. Figure 3-50 illustrates a potential methodology for approximating seasonal flow needs based on the ecological risk levels developed for various attributes as part of the WFET ecological risk mapping. The following attributes were considered when developing this approach:

- Trout
- Riparian
- Recreation

The trout ecological risk metric is based on August and September mean flow conditions. The risk levels and range of flows associated with these levels can be used to estimate a range of August and September flows. For riparian conditions, the ecological risk metric is based on a one-day peak flow which usually occurs in late spring or early summer.

So, for this example, a range of June flows could be estimated based on the ecological risk levels for riparian areas. For May and July, a range of flows could be generated by considering the recreational flow metric. In this example, recreation season was assumed to occur during the months of May through August. However, the June riparian flows and August trout flows were assumed to take precedence over recreation in these months. For all other months where there is not an attribute with an ecological flow metric association, the CWCB's decreed instream flow water rights could be used to approximate base flow conditions. This example focuses on attributes in the Roaring Fork Watershed. Future applications using this approach outside of the Roaring Fork Watershed will have to consider other attributes based to develop similar risk-based seasonal flows.



3-50. Methodology for Developing Range of Seasonal Flow Conditions Based on WFET Ecological Risk Levels

An example range of flow conditions that may be generated using the approach described above is shown in Figure 3-51. A range of flow conditions was developed for the node located at the confluence of the Roaring Fork River and Castle Creek. The range of seasonal flows associated with the ecological risk levels for the attributes described above are shown in the red, orange, yellow, and green lines on the figure. These risk levels are the same as in the ecological risk maps discussed previously where red indicates a higher risk and green represents a lower risk. For comparison purposes, the average monthly baseline and average monthly existing flows are also shown on Figure 3-51. These flows represent average conditions for the Upper Colorado Model period of record (1975 – 2005). This figure is comparable to the ecological risk mapping for this area where the riparian attribute had a higher ecological risk, the recreation attribute was at a higher risk level, and the trout attribute shows a lower minimal ecological risk. The range of seasonal flows shown in this figure could be refined and used during the water availability modeling conducted as part of the Colorado River Supply Availability Study to represent a range of demands representing environmental and recreational needs in the Colorado River basin. Further refinement of this approach is needed to include intra-year and inter-year variations.

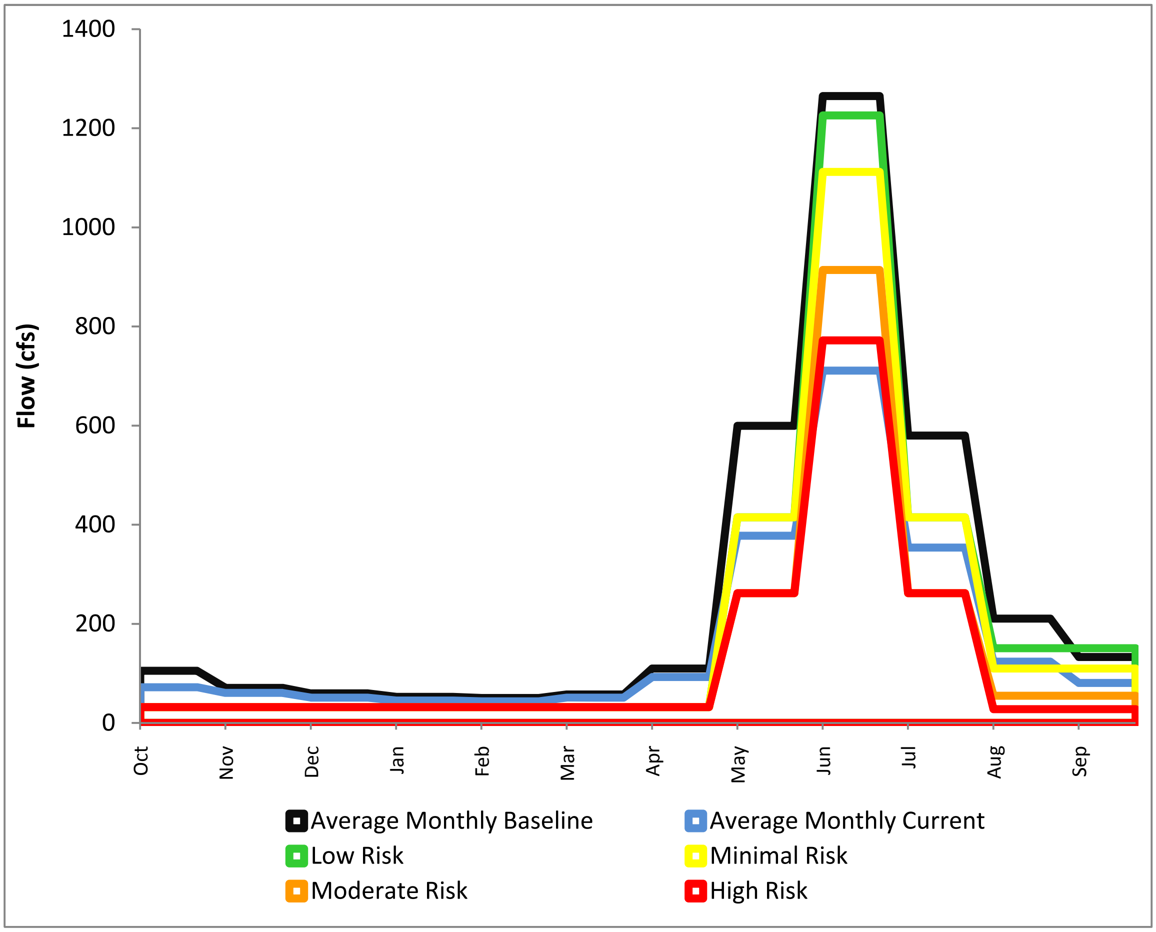
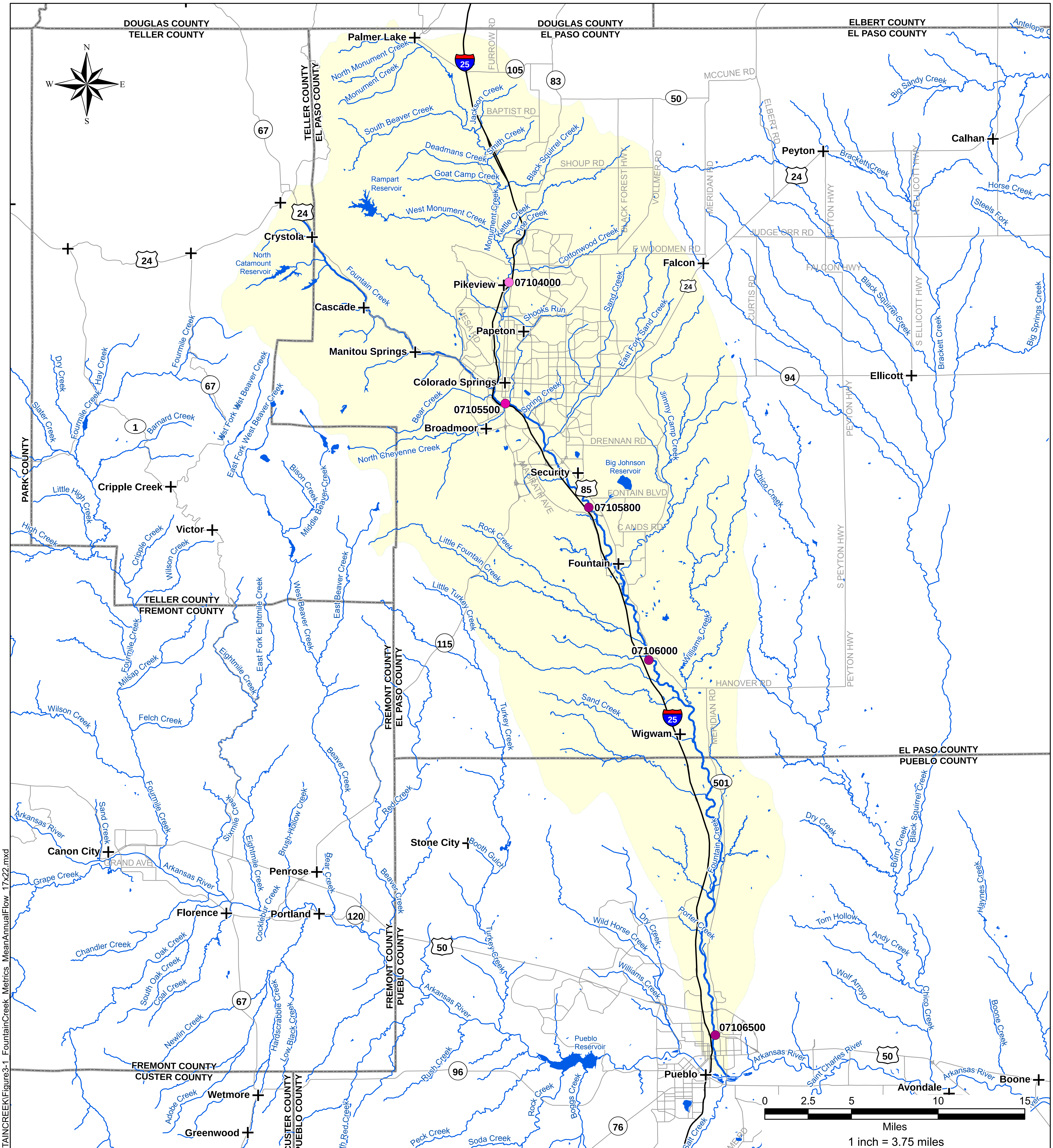


Figure 3-51. Example of Range of Risk-Based Seasonal Flow Demands for Nonconsumptive Attributes in the Roaring Fork River at Confluence of Castle Creek



Legend

Mean Annual Flow Percent Increase

- < 30%
- 30 to 150%
- > 150%
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-1
Fountain Creek Watershed
Flow Evaluation Tool

Mean Annual Flow Percent Increase from Baseline to Existing Conditions

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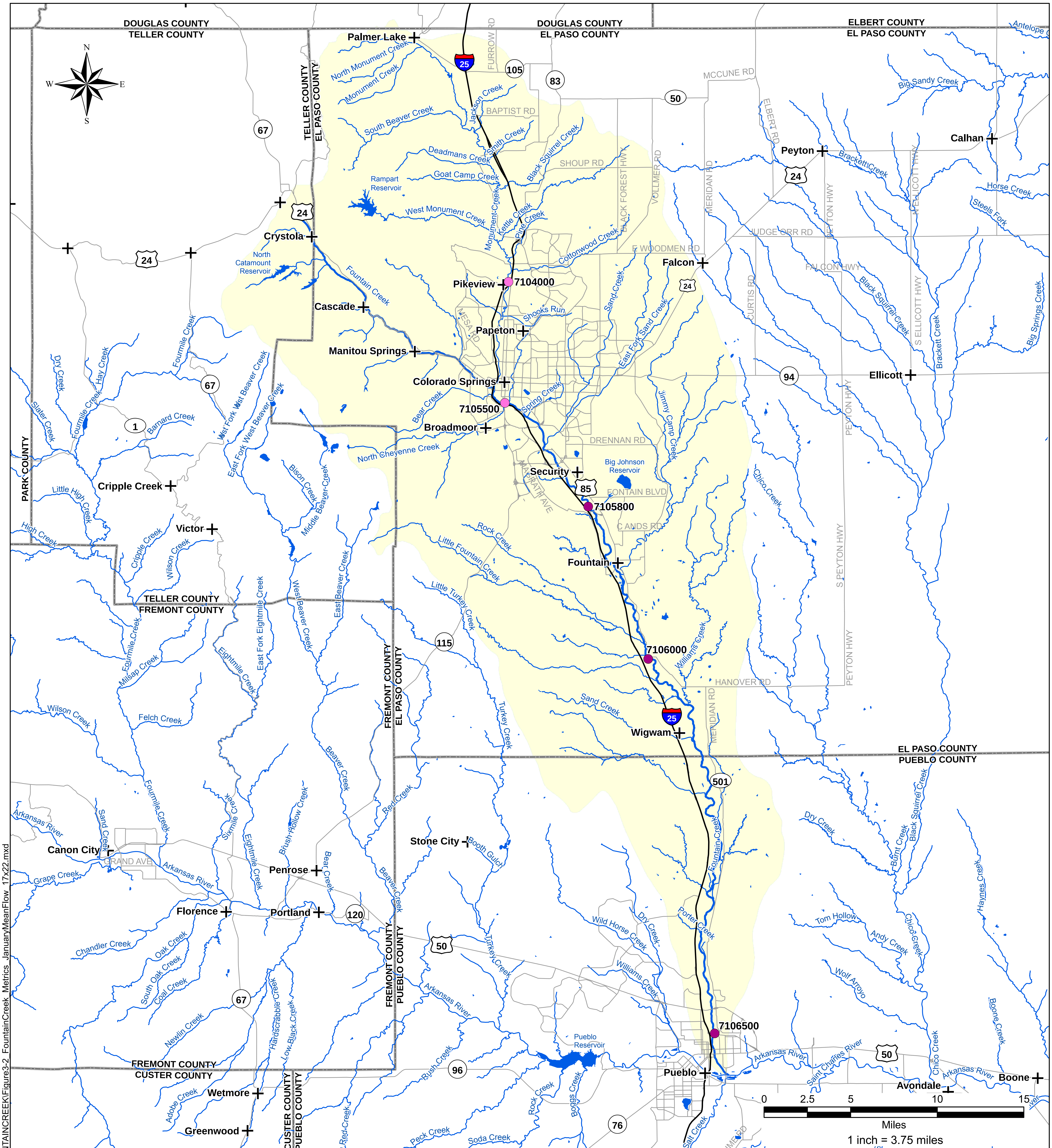
Document: Z:\SWSI\NONCONSUMPTIVE\WFET\MXD\FOUNTAINCREEK\Figure3-1_FountainCreek_Metrics_MeanAnnualFlow_17x22.mxd



Legend

Mean January Flow Percent Increase

- < 75%
- 75 to 150%
- > 150%
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-2
Fountain Creek Watershed
Flow Evaluation Tool

Mean January Flow Percent Increase from Baseline to Existing Conditions

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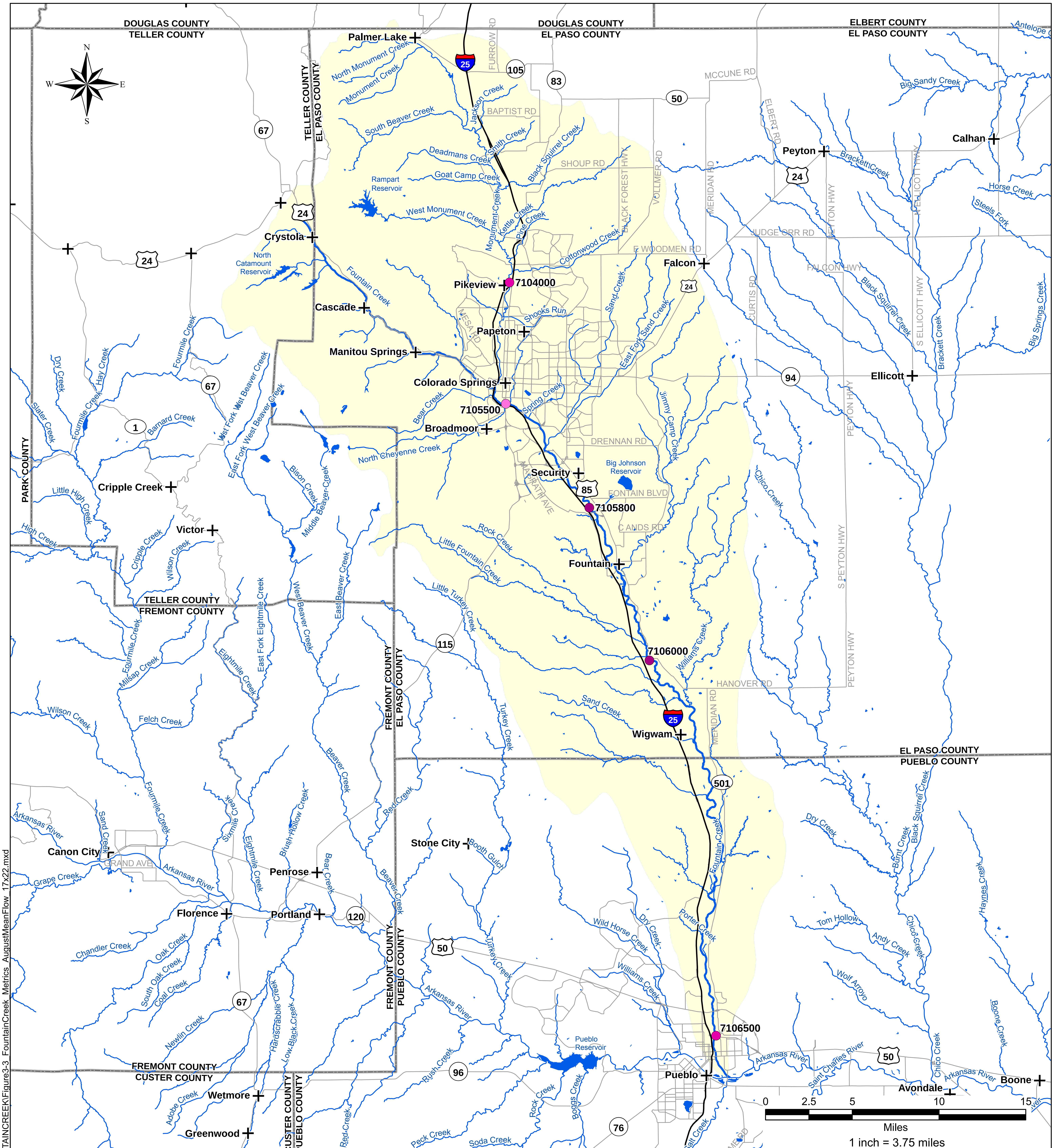

Legend

Mean August Flow Percent Increase

- < 50%
- 50 to 150%
- > 150%
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-3
Fountain Creek Watershed
Flow Evaluation Tool

Mean August Flow Percent Increase from Baseline to Existing Conditions

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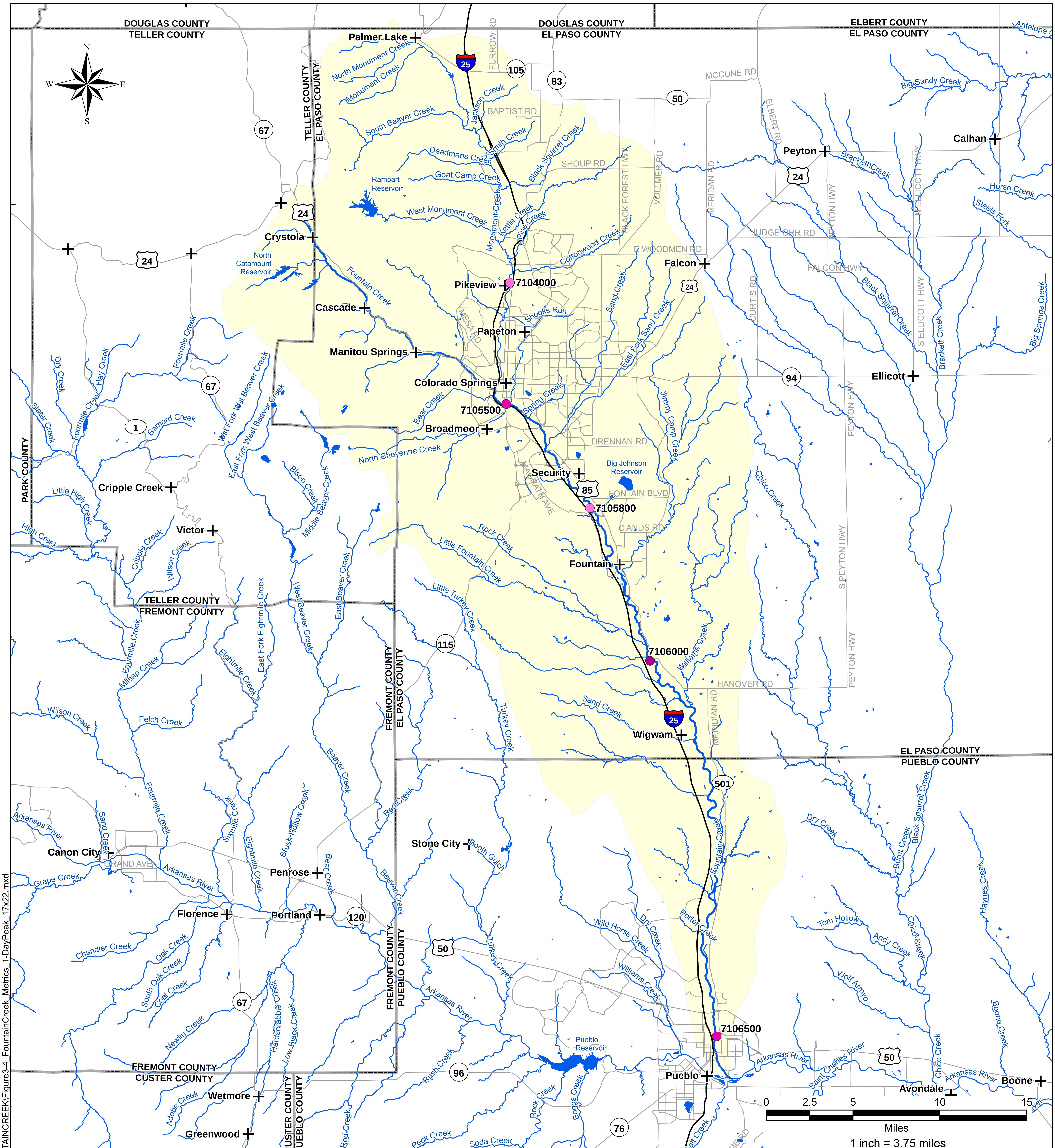

Legend

1-Day Peak Flow Percent Increase

- < 25%
- 25 to 75 %
- > 75%
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-4
Fountain Creek Watershed
Flow Evaluation Tool

1-Day Peak Flow Percent Increase
from Baseline to Existing Conditions

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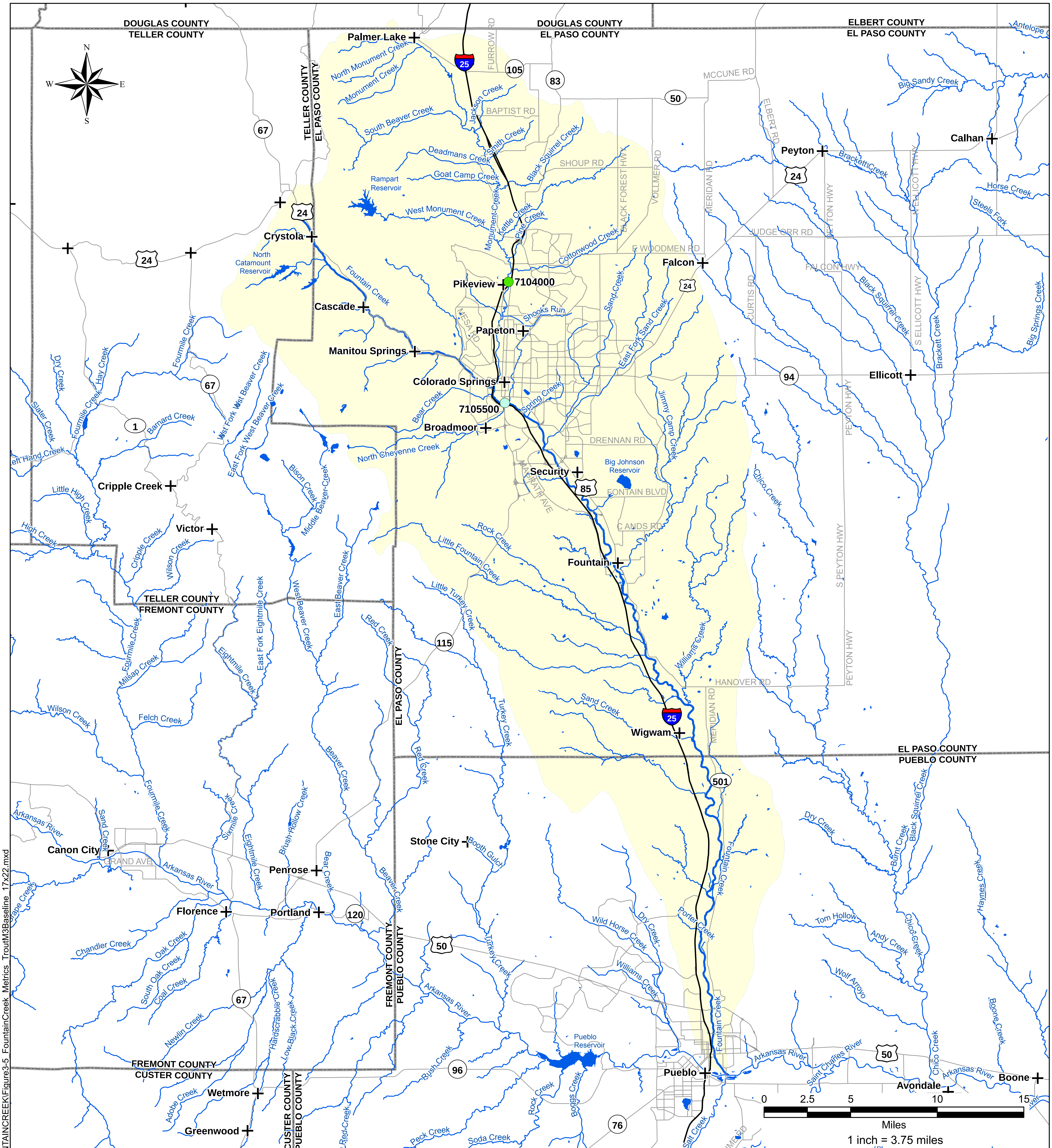

Legend

Trout Method 3 Baseline Conditions

Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Signficant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-5
Fountain Creek Watershed
Flow Evaluation Tool

Trout Baseline Conditions
Risk Mapping

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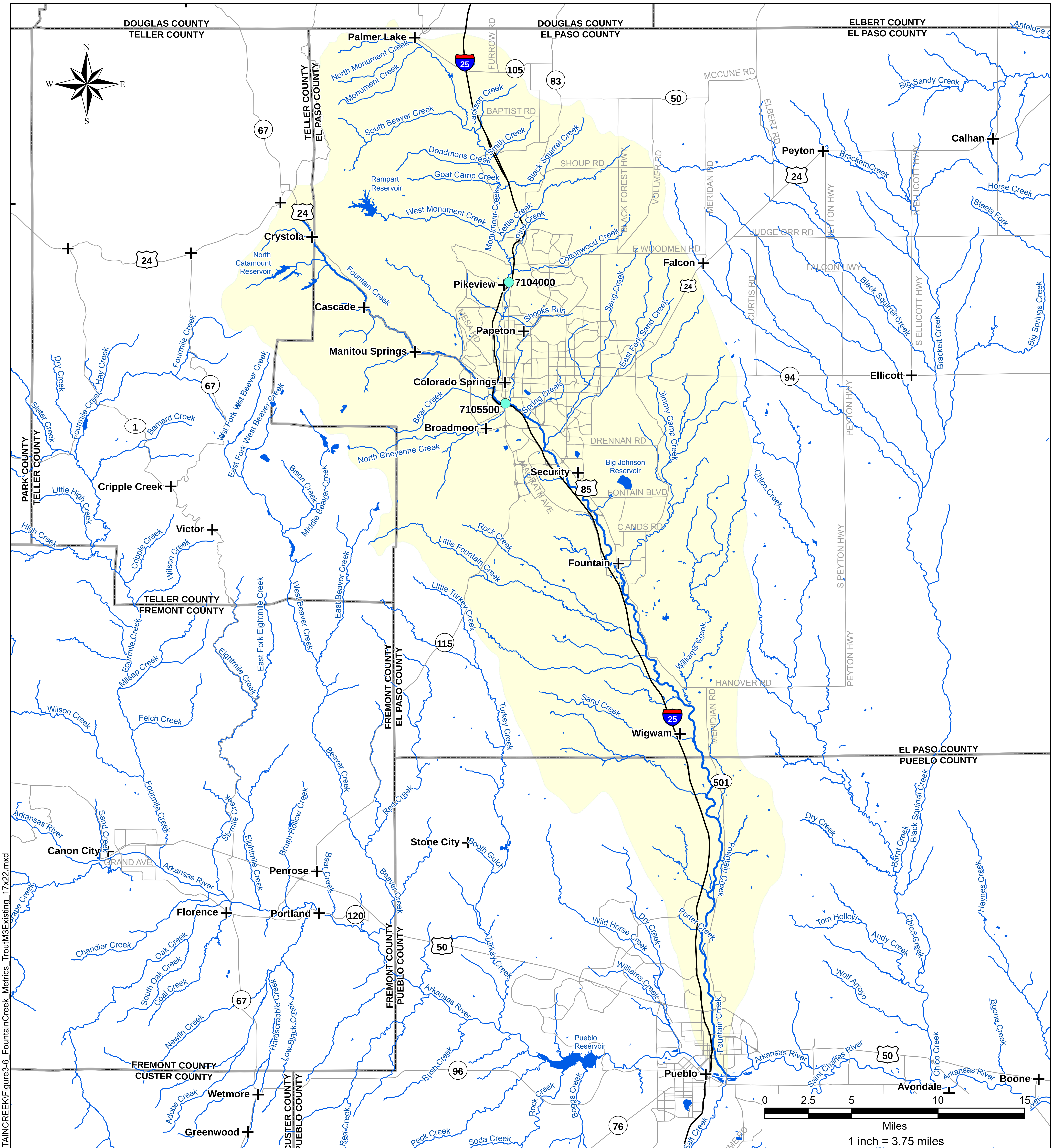

Legend

Trout Method 3 Existing Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55%- Low Ecological Risk

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-6
Fountain Creek Watershed
Flow Evaluation Tool

Trout Existing Conditions
Risk Mapping

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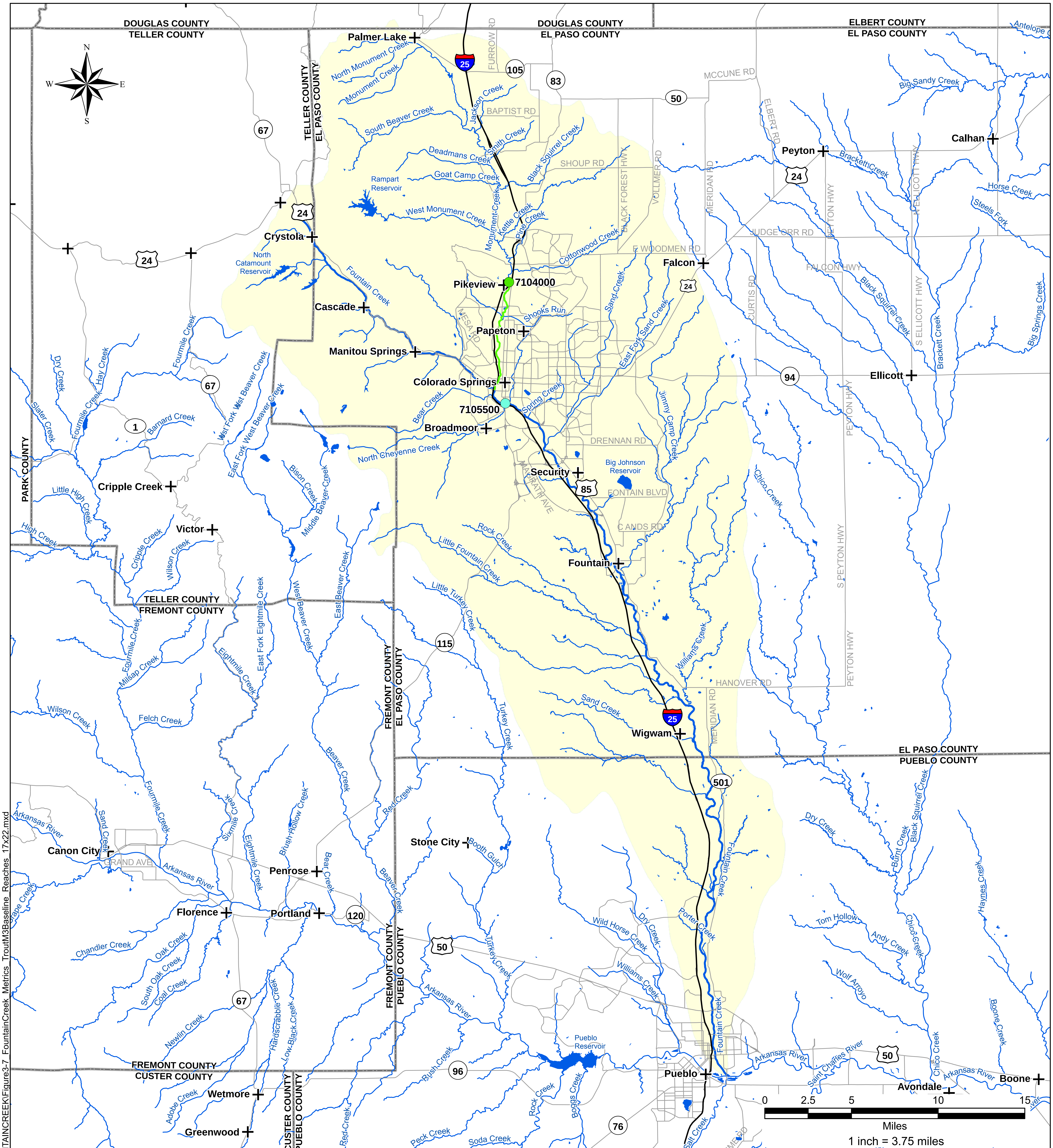

Legend

Trout Method 3 Baseline Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-7
Fountain Creek Watershed
Flow Evaluation Tool

Trout Baseline
Conditions Risk Reach Mapping

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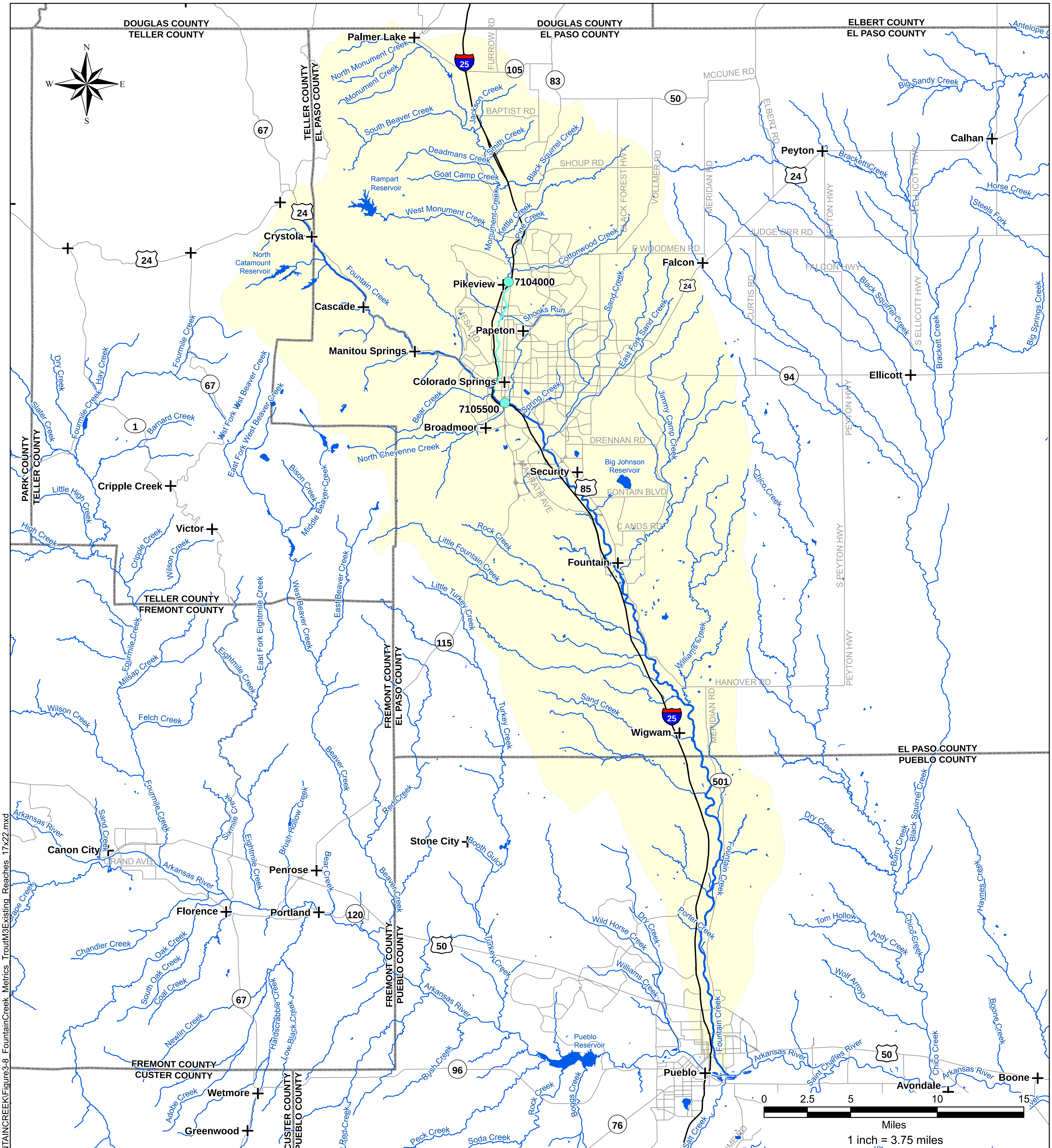

Legend

Trout Method 3 Existing Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-8
Fountain Creek Watershed Flow Evaluation Tool

Trout Existing Conditions Risk Reach Mapping

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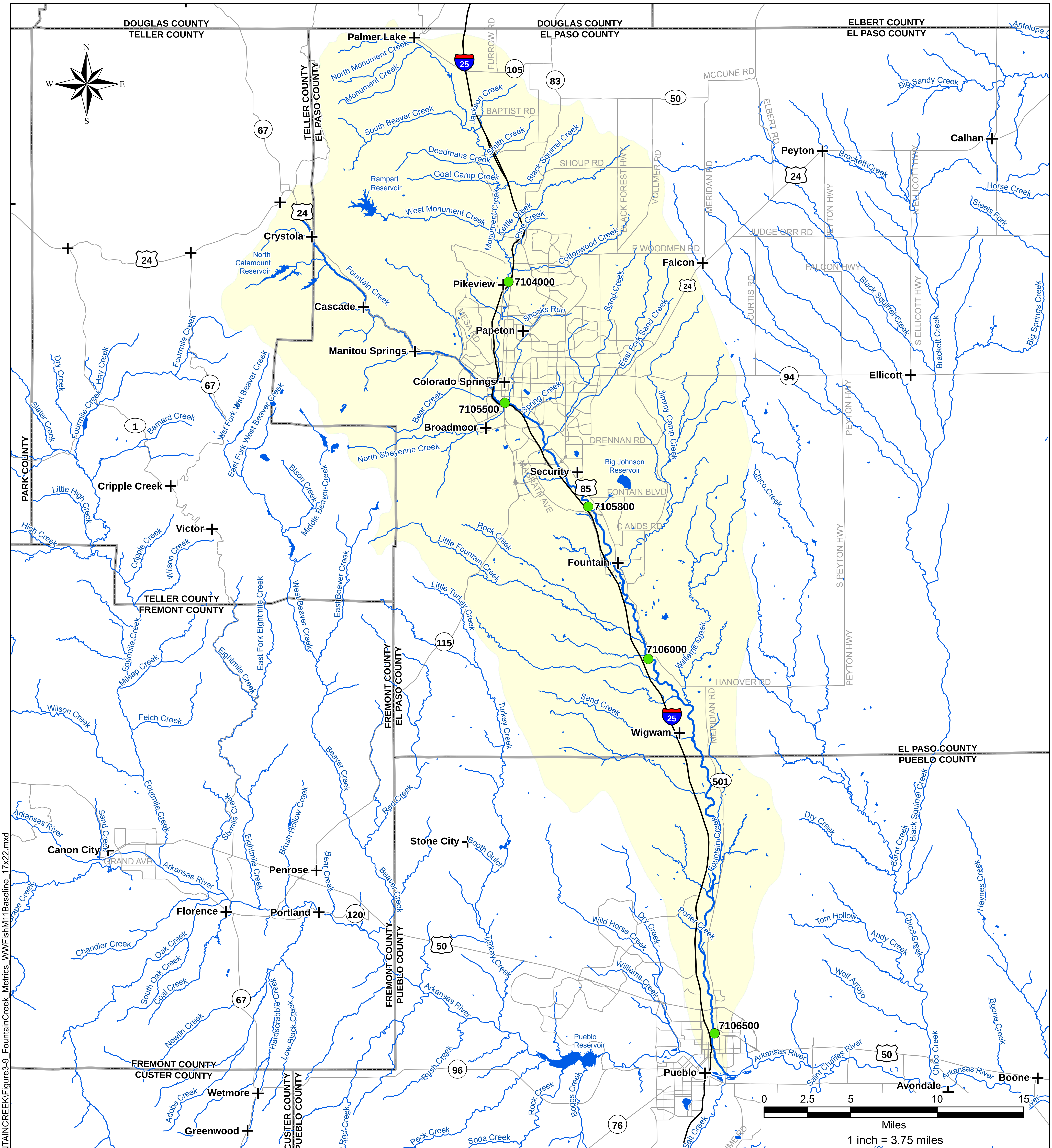

Legend

Warm Water Fish Method 11
Baseline Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 30% - Moderate Ecological Risk
- 31 - 40% - Minimal Ecological Risk
- > 40% - Low Ecological Risk
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-9
Fountain Creek Watershed
Flow Evaluation Tool

Warm Water Fish
Baseline Conditions Risk Mapping

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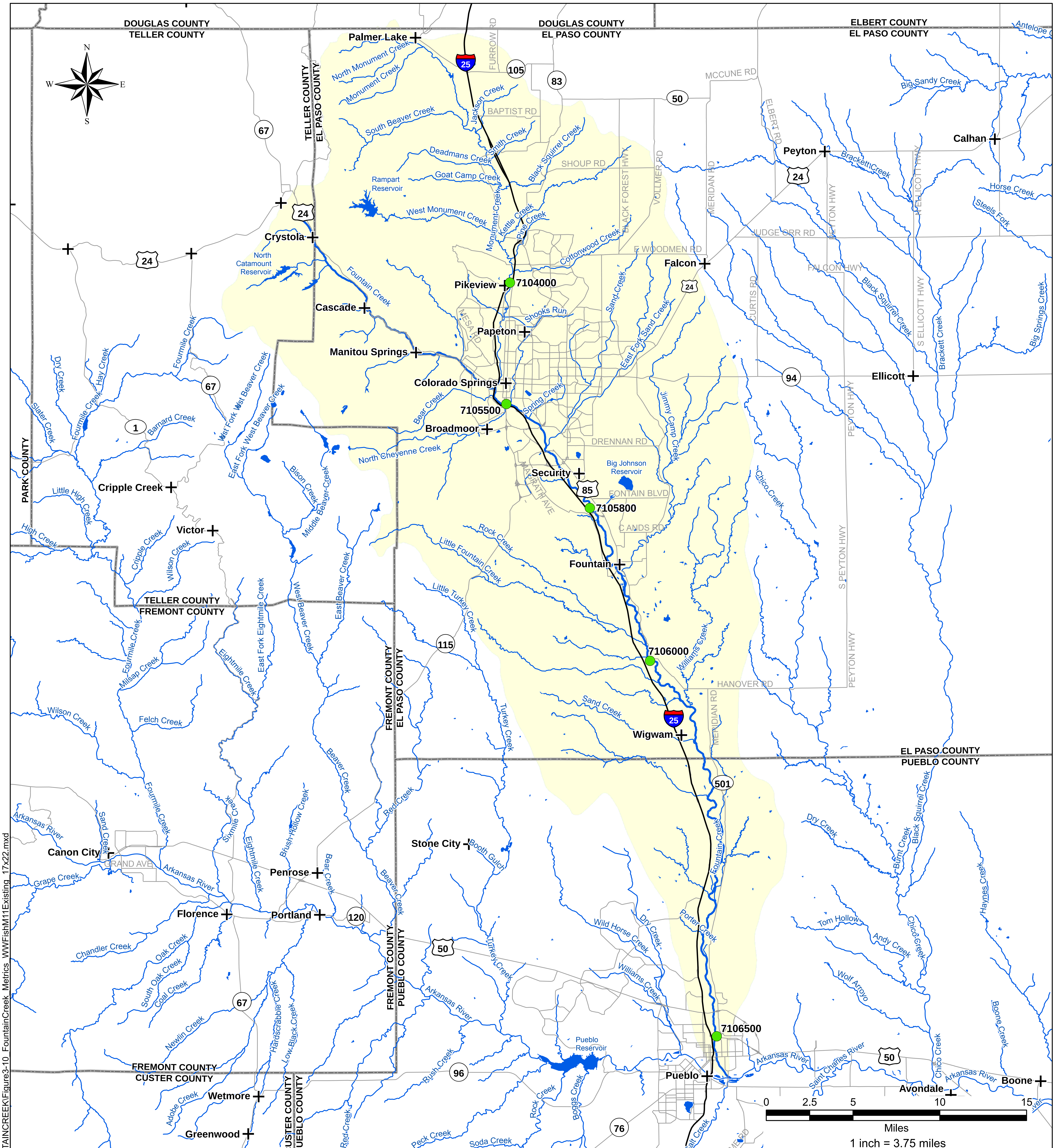

Legend

Warm Water Fish Method 11

Baseline Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 30% - Moderate Ecological Risk
- 31 - 40% - Minimal Ecological Risk
- \> 40% - Low Ecological Risk
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-10
Fountain Creek Watershed
Flow Evaluation Tool

Warm Water Fish
Existing Conditions Risk Mapping

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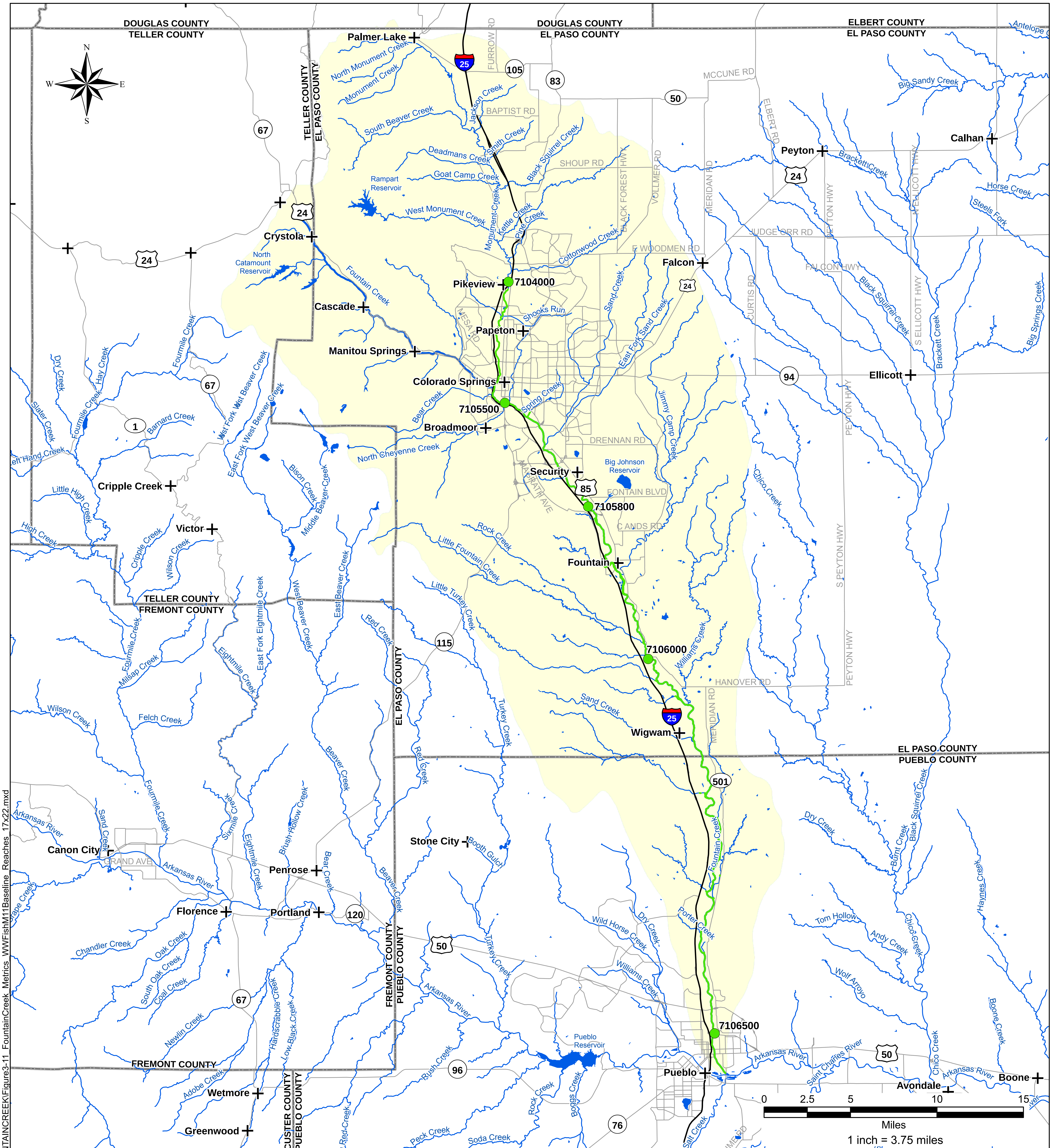

Legend

Warm Water Fish Method 11
Baseline Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 30% - Moderate Ecological Risk
- 31 - 40% - Minimal Ecological Risk
- > 40% - Low Ecological Risk
- < 10% - High Ecological Risk
- 10 - 30% - Moderate Ecological Risk
- 31 - 40 % - Minimal Ecological Risk
- > 40% - Low Ecological Risk
- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-11
Fountain Creek Watershed
Flow Evaluation Tool

Warm Water Fish
Baseline Conditions Risk Reach Mapping

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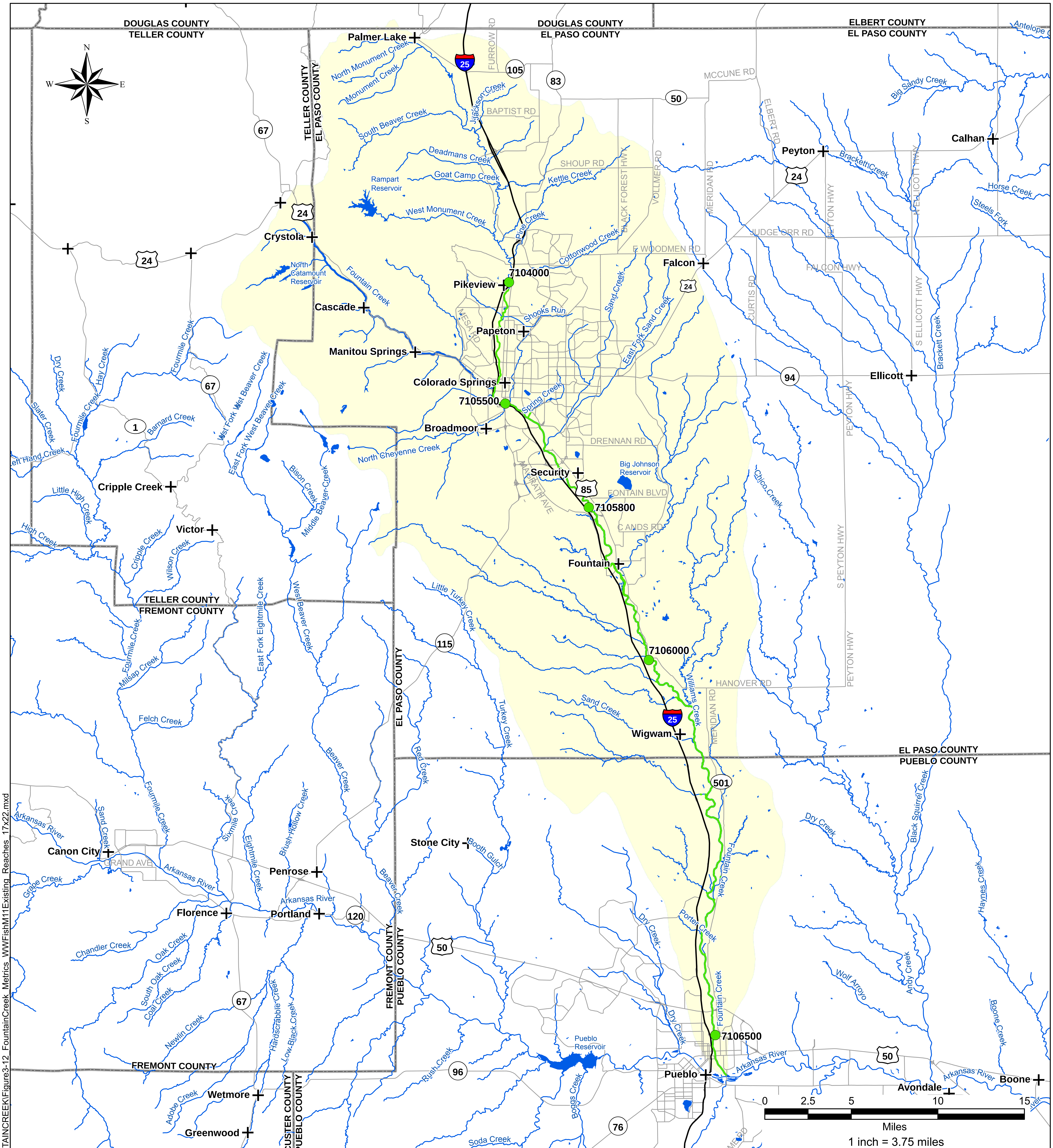

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Legend

Warmwater Fish Method 11
Baseline Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 30% - Moderate Ecological Risk
- 31 - 40% - Minimal Ecological Risk
- > 40% - Low Ecological Risk
- < 10% - High Ecological Risk
- 10 - 30% - Moderate Ecological Risk
- 31 - 40% - Minimal Ecological Risk
- > 40% - Low Ecological Risk

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-12
Fountain Creek Watershed
Flow Evaluation Tool

Warm Water Fish
Existing Conditions Risk Reach Mapping

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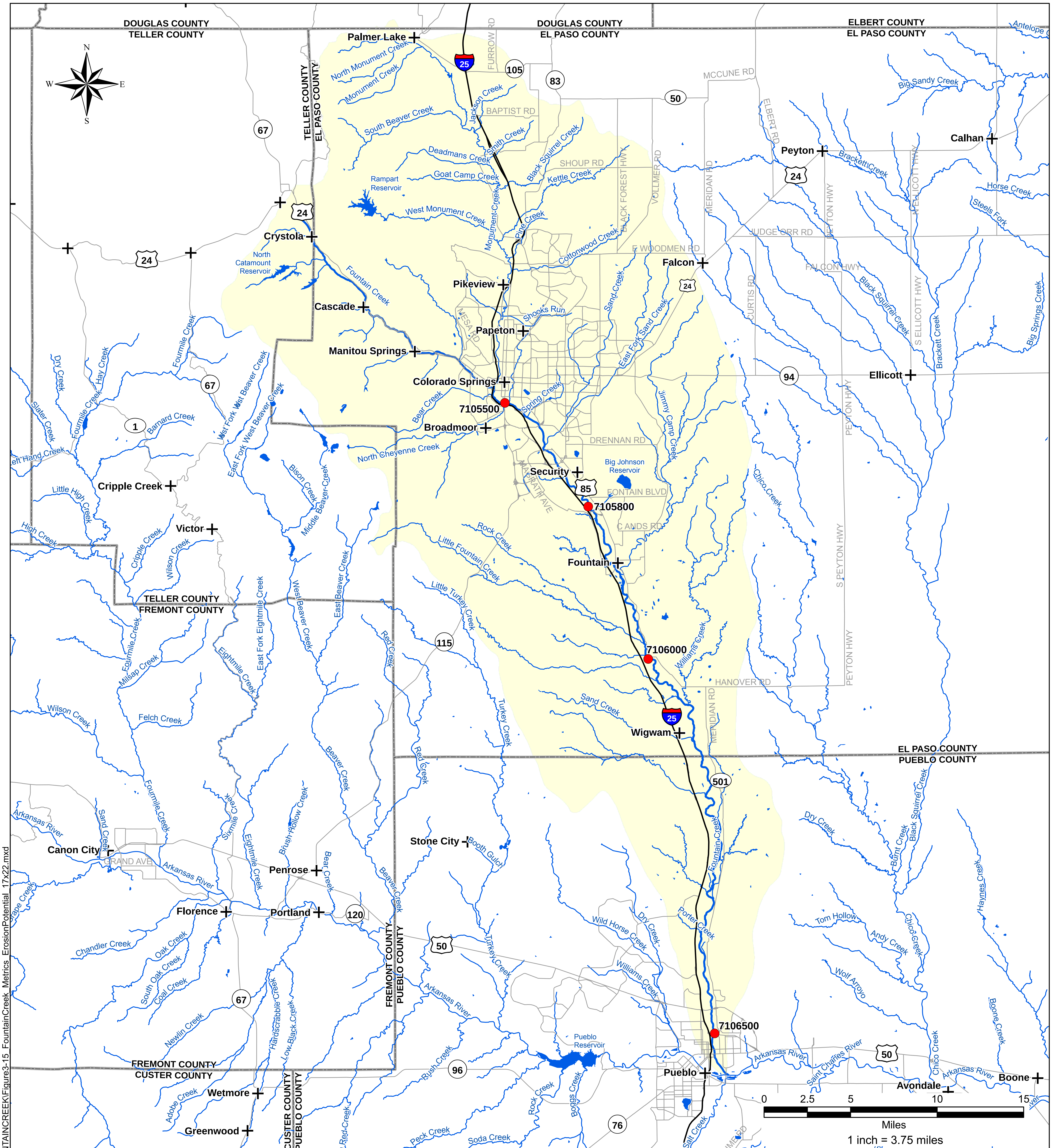

Legend

Erosion Potential

- 1 - 2 - Low Erosion Potential
- 2 - 4
- > 4 - High Erosion Potential

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-15
Fountain Creek Watershed Flow Evaluation Tool

Erosion Potential Risk Mapping

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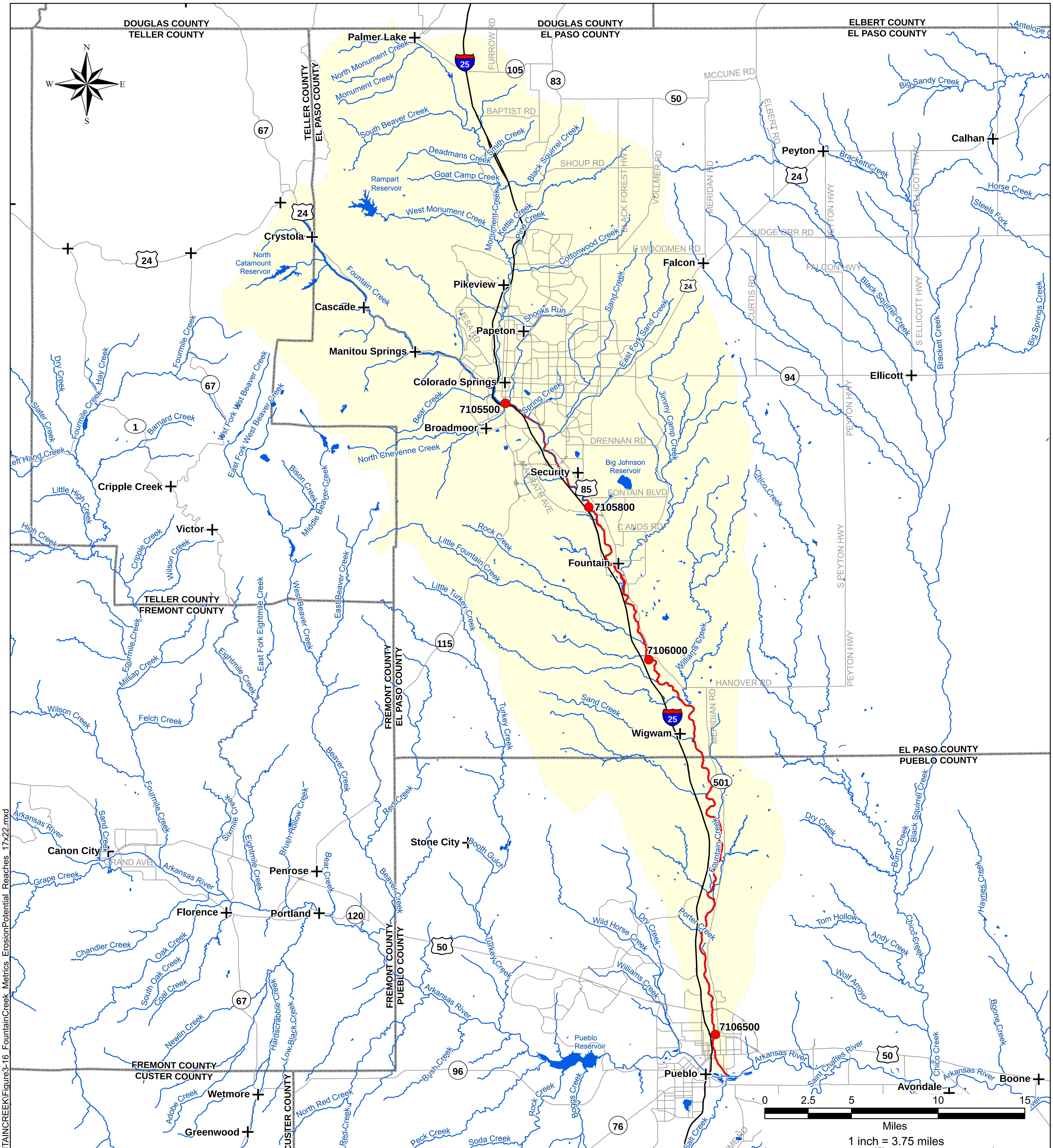

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Legend

Erosion Potential

- 1 - 2 - Low Erosion Potential
- 2 - 4
- > 4 - High Erosion Potential
- 1 - 2 - Low Erosion Potential
- 2 - 4
- > 4 - High Erosion Potential

- Lakes and Reservoirs
- Streams and Rivers
- Roads
- Highways
- Cities and Towns
- Counties
- Watershed

Figure 3-16
Fountain Creek Watershed
Flow Evaluation Tool

Erosion Potential Risk Reach Mapping

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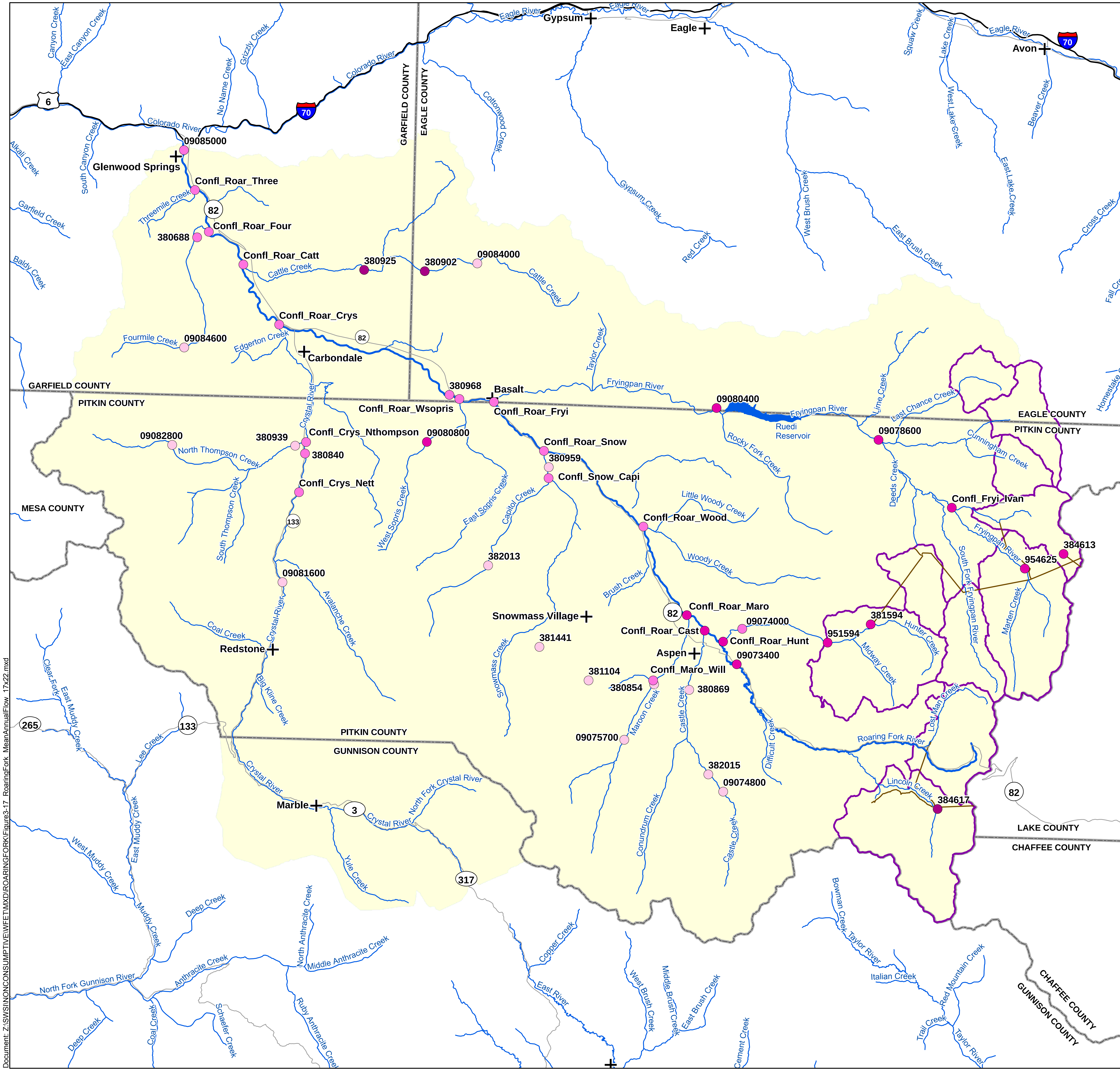

Legend

Mean Annual Flow Percent Decrease

- < 10%
- 10 to 25%
- 26 to 50%
- > 50%
- Transbasin Diversion Areas
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

N E S W

0 2 4 8

Miles

1 inch = 3.25 miles

Figure 3-17
Roaring Fork Watershed
Flow Evaluation Tool

Mean Annual Flow Percent Decrease from Baseline to Existing Conditions

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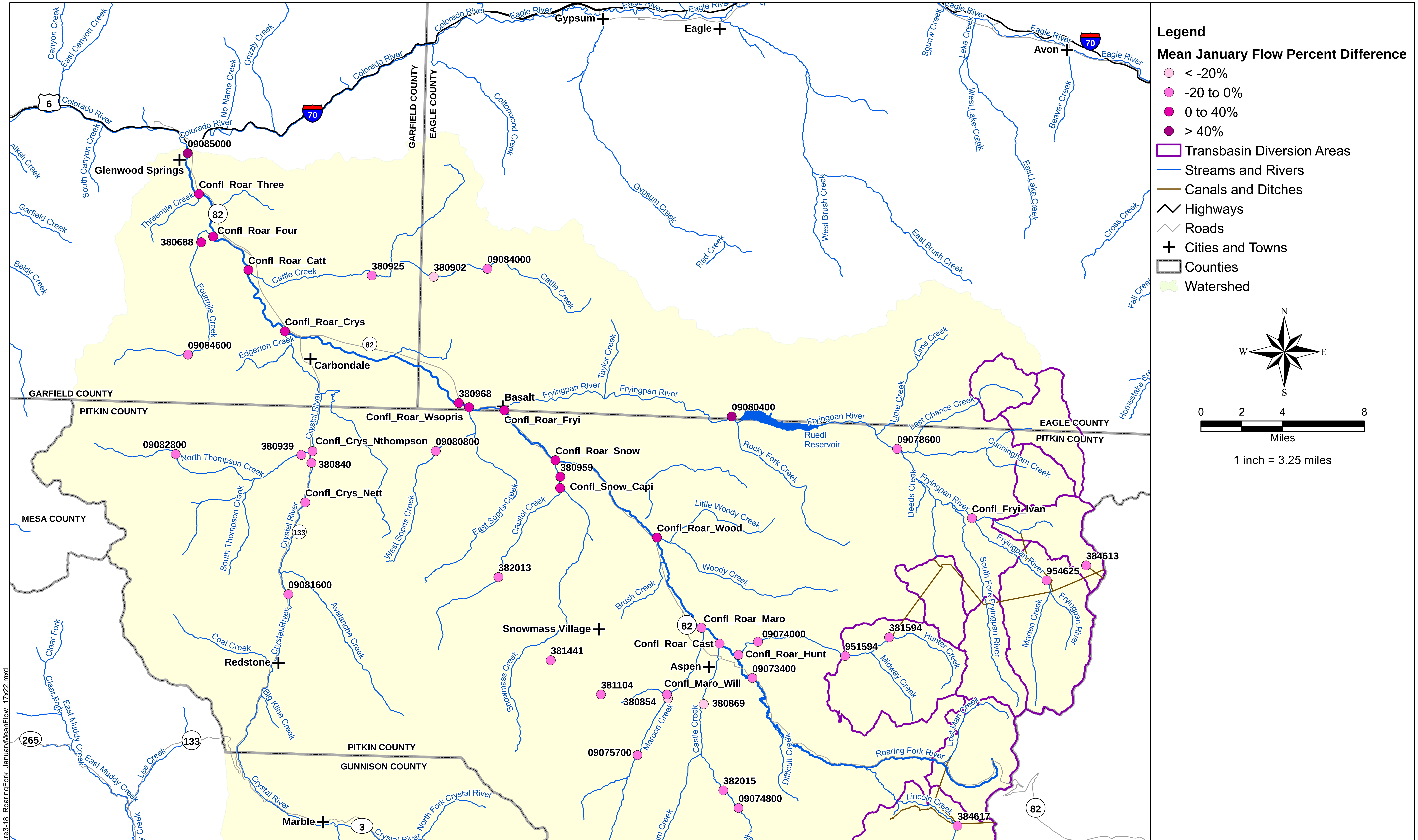

Figure 3-18
Roaring Fork Watershed
Flow Evaluation Tool

Mean January Flow Percent Difference from Baseline to Existing Conditions

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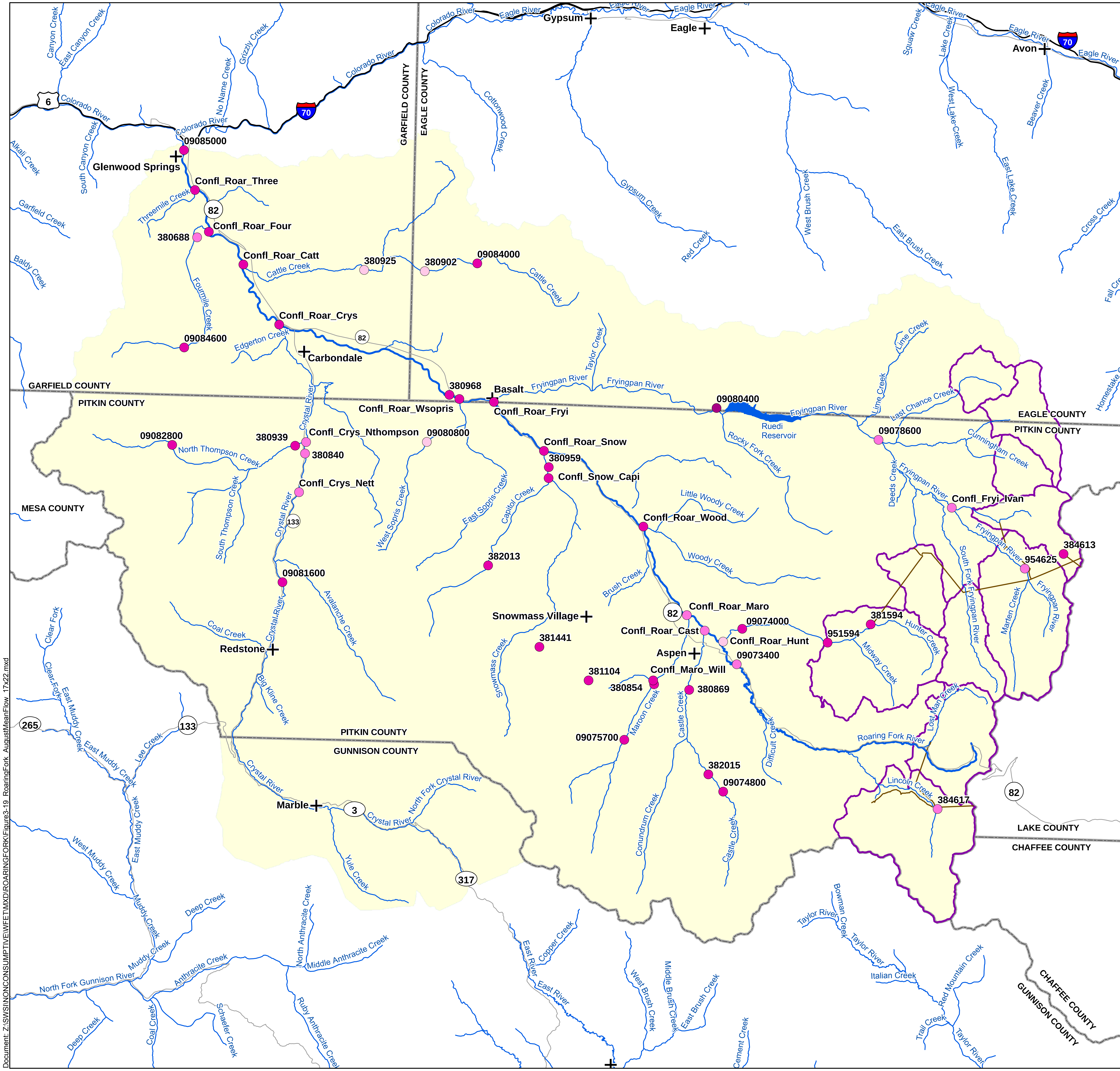

Figure 3-19
Roaring Fork Watershed
Flow Evaluation Tool

Mean August Flow Percent Difference from Baseline to Existing Conditions

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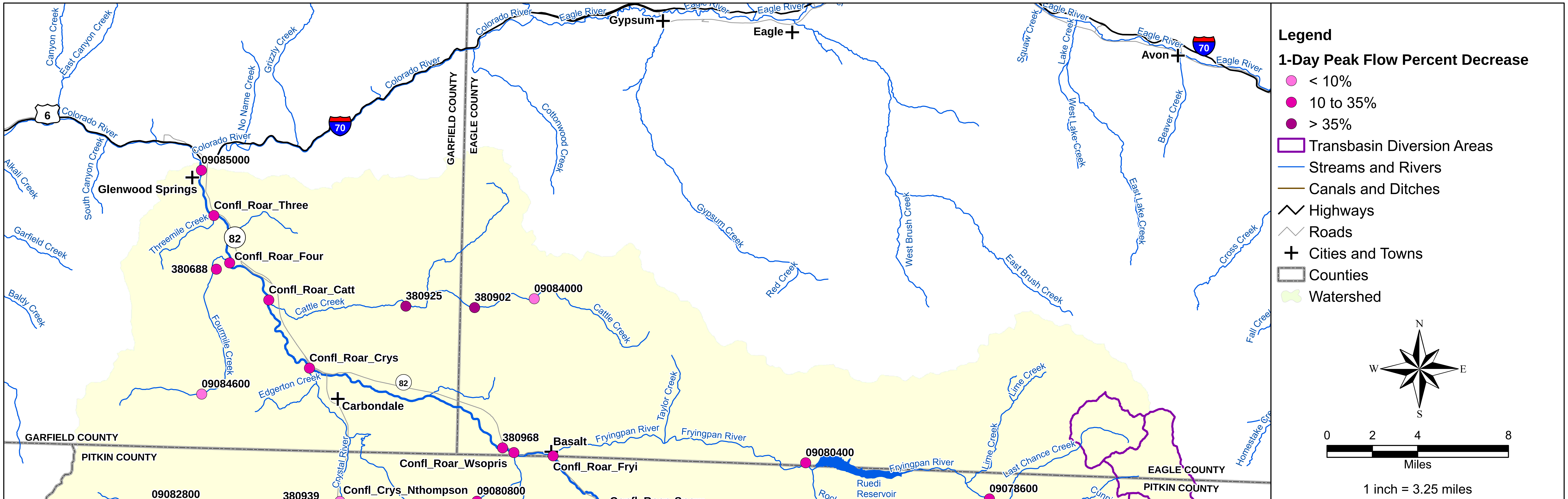

Figure 3-20
Roaring Fork Watershed Flow Evaluation Tool

1-Day Peak Flow Percent Decrease from Baseline to Existing Conditions

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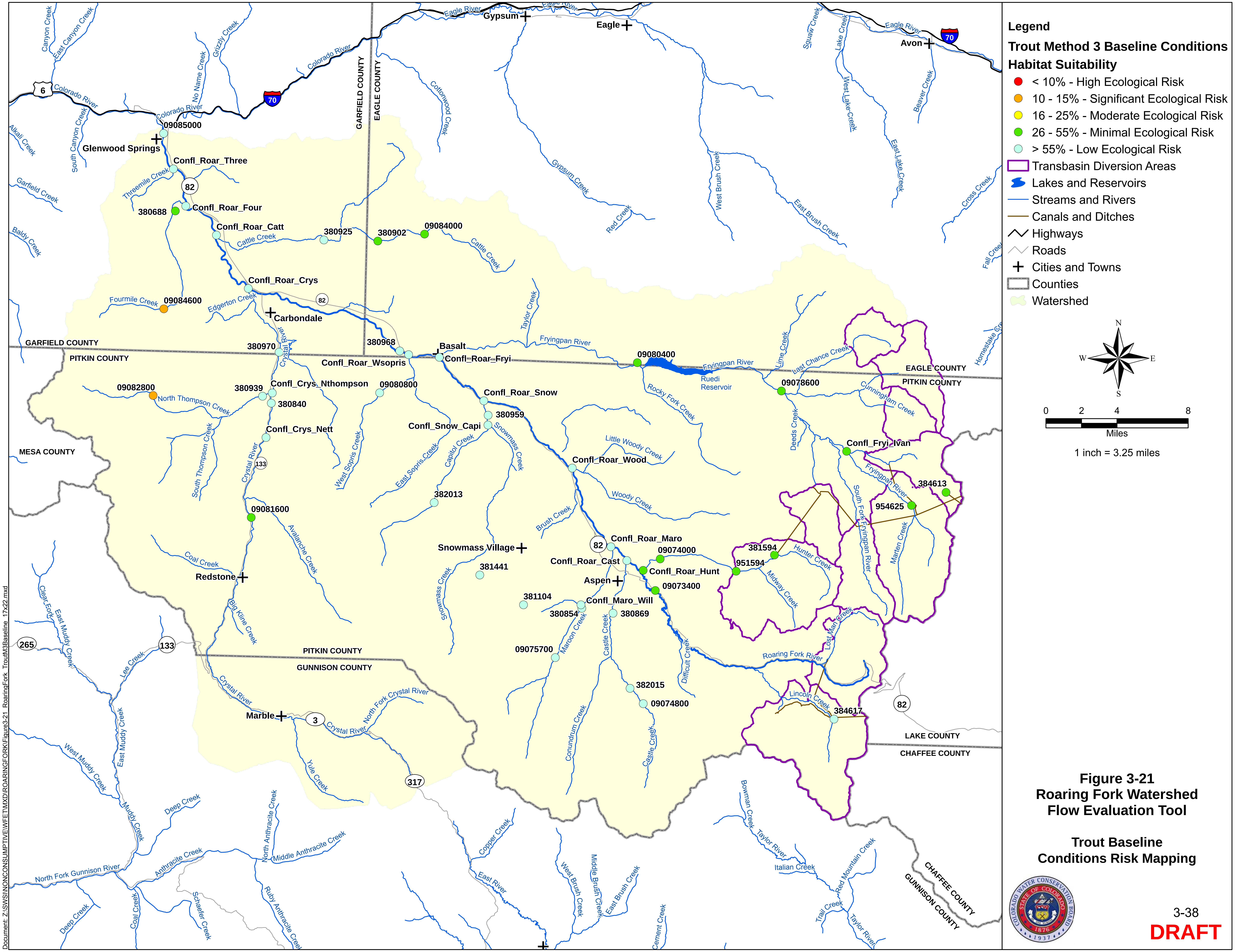
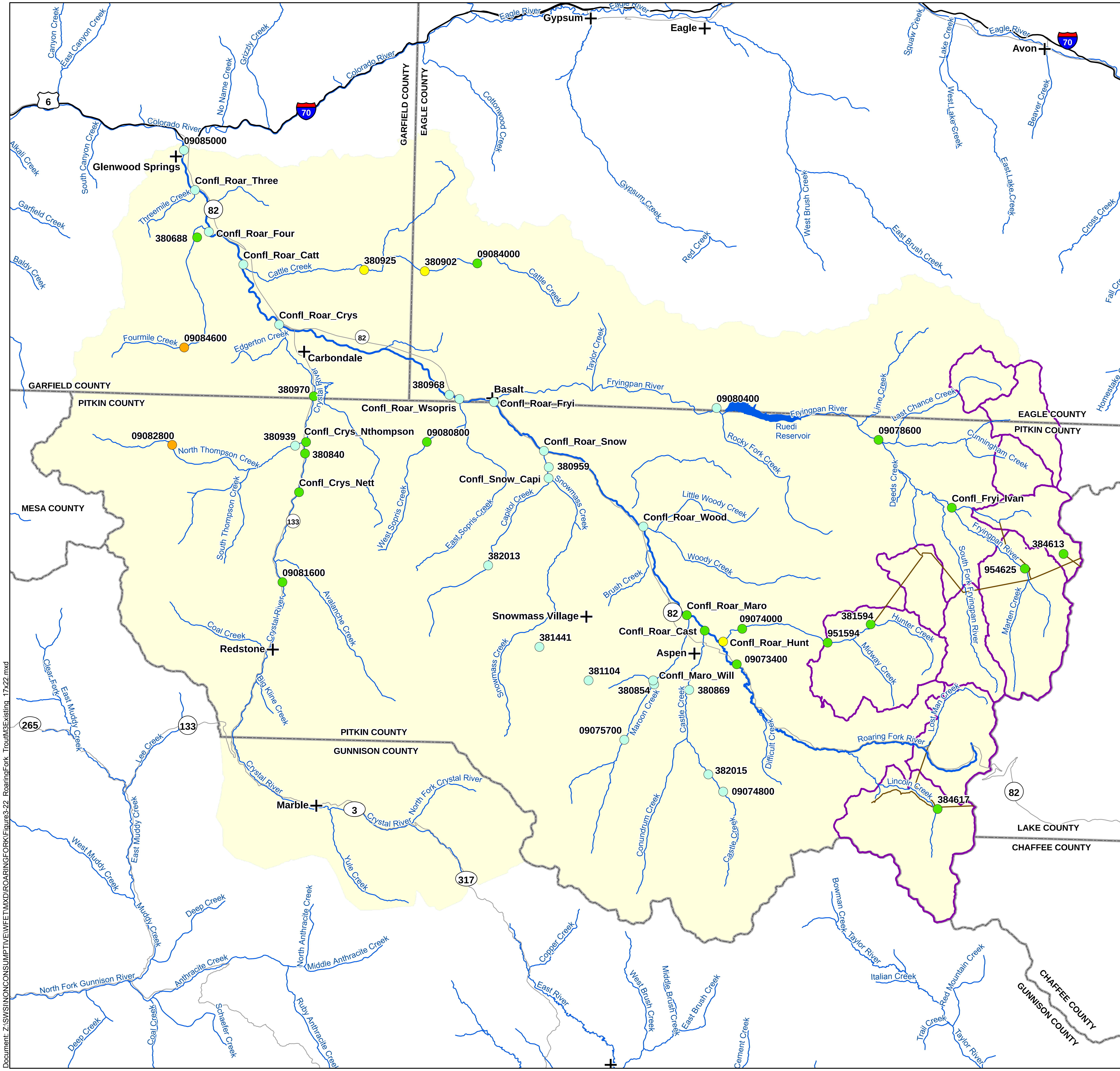


Figure 3-21
Roaring Fork Watershed
Flow Evaluation Tool

Trout Baseline
Conditions Risk Mapping

DRAFT



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Legend

Trout Method 3 Existing Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- Transbasin Diversion Areas
- Lakes and Reservoirs
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

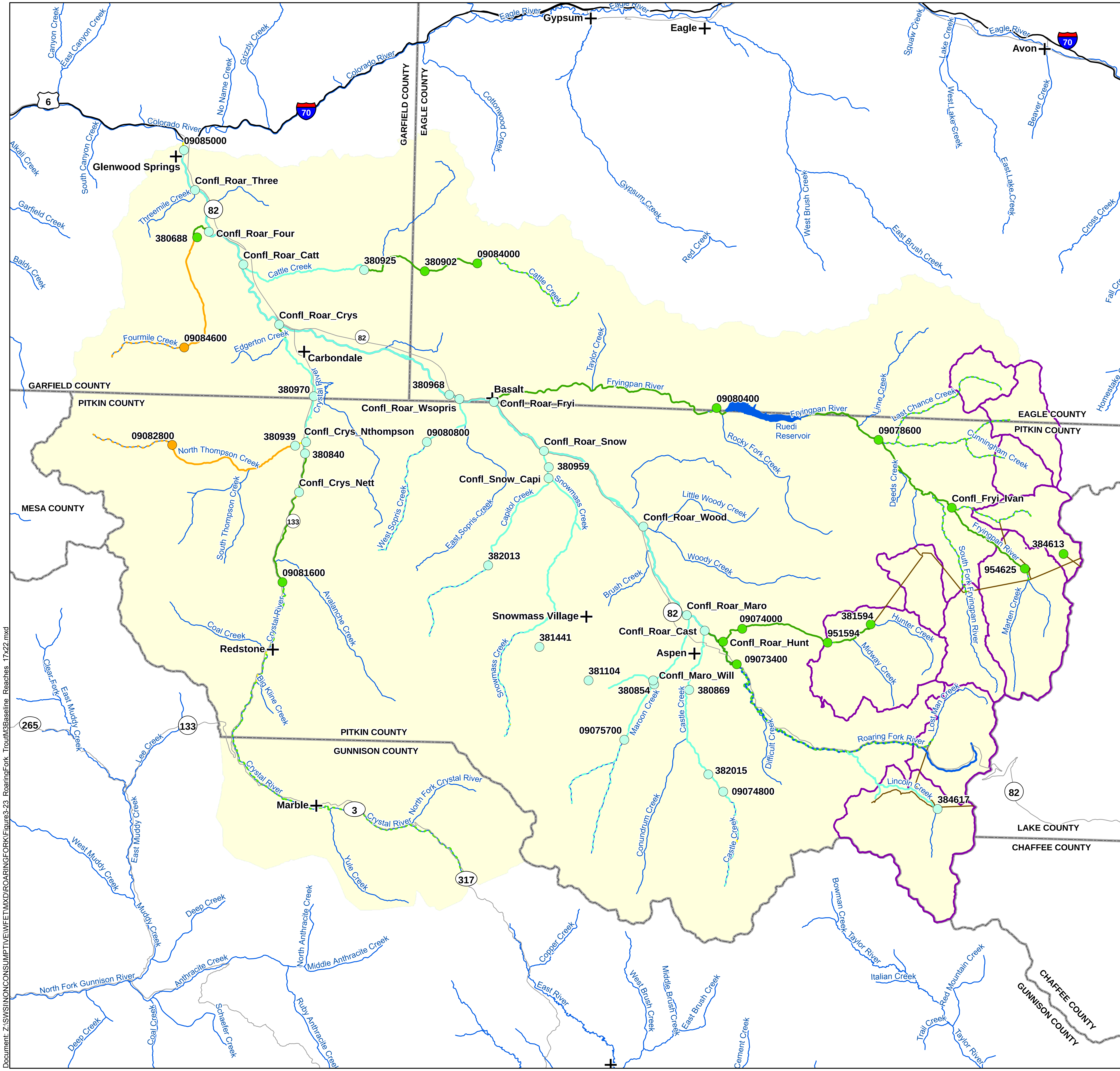
0 2 4 8
Miles

1 inch = 3.25 miles

Figure 3-22
Roaring Fork Watershed
Flow Evaluation Tool

Trout Existing
Conditions Risk Mapping

DRAFT



Legend

Trout Method 3 Baseline Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- Transbasin Diversion Areas
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

0 2 4 8
Miles
1 inch = 3.25 miles

Figure 3-23
Roaring Fork Watershed
Flow Evaluation Tool

Trout Baseline
Conditions Risk
Reach Mapping

DRAFT

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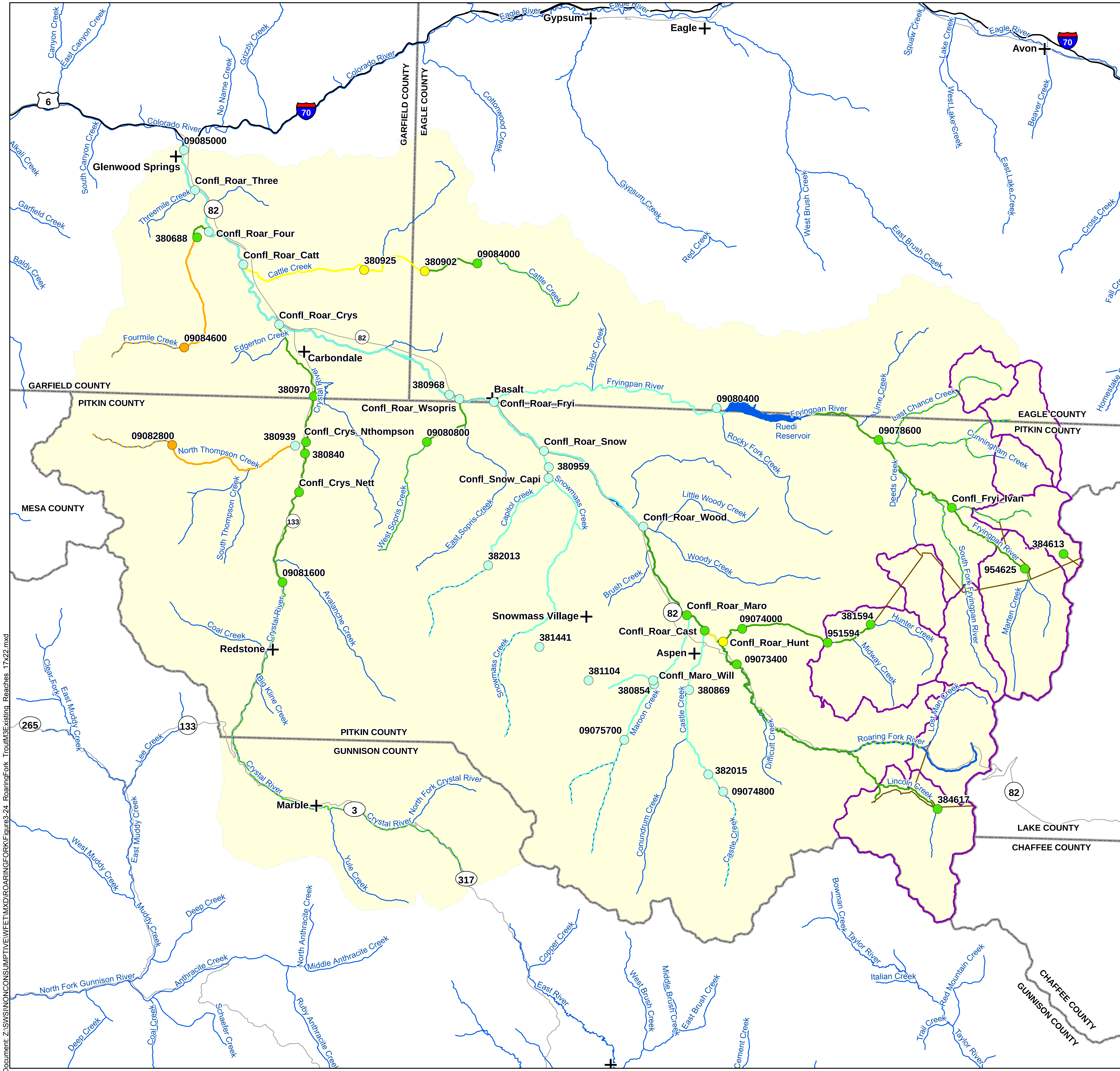

Legend

Trout Method 3 Existing Conditions Habitat Suitability

- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- < 10% - High Ecological Risk
- 10 - 15% - Significant Ecological Risk
- 16 - 25% - Moderate Ecological Risk
- 26 - 55% - Minimal Ecological Risk
- > 55% - Low Ecological Risk
- Transbasin Diversion Areas
- Lakes and Reservoirs
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

0 2 4 8
Miles

1 inch = 3.25 miles

Figure 3-24
Roaring Fork Watershed
Flow Evaluation Tool

Trout Existing Conditions Risk Reach Mapping

DRAFT

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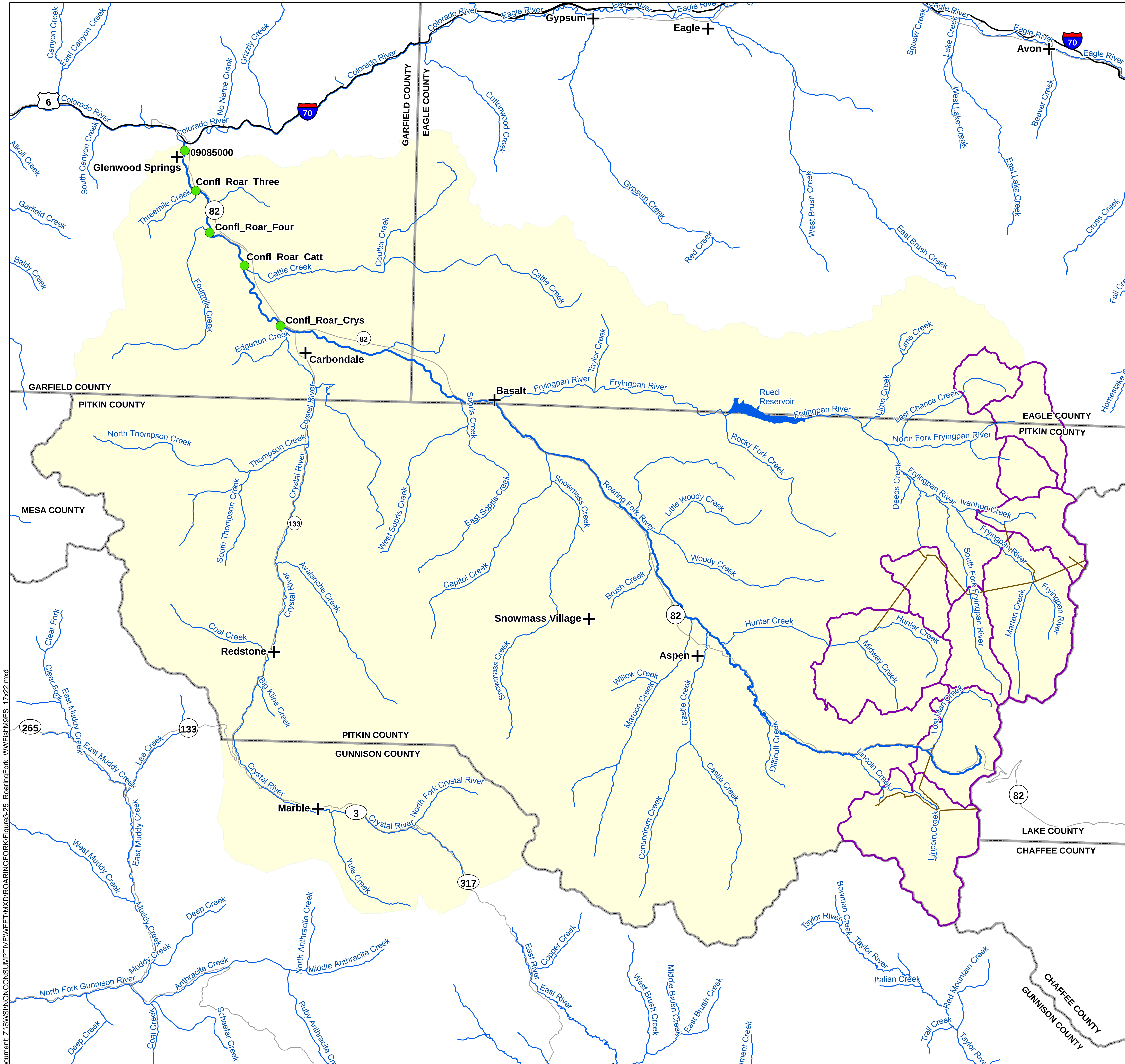

Legend

Warm Water Fish Method 9
Flannelmouth Sucker
Percent Reduction in Biomass
from Baseline to Existing Conditions

- < 10% - Low Ecological Risk
- 10 - 25% - Minimal Ecological Risk
- 26 - 50% - Moderate Ecological Risk
- ≥ 51% - High Ecological Risk
- Transbasin Diversion Areas
- Lakes and Reservoirs
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

N
W E
S

0 2 4 8
Miles

1 inch = 3.25 miles

Figure 3-25
Roaring Fork Watershed
Flow Evaluation Tool

Flannelmouth Sucker Risk Mapping

COLORADO WATER CONSERVATION BOARD STATE OF COLORADO 1876 1937

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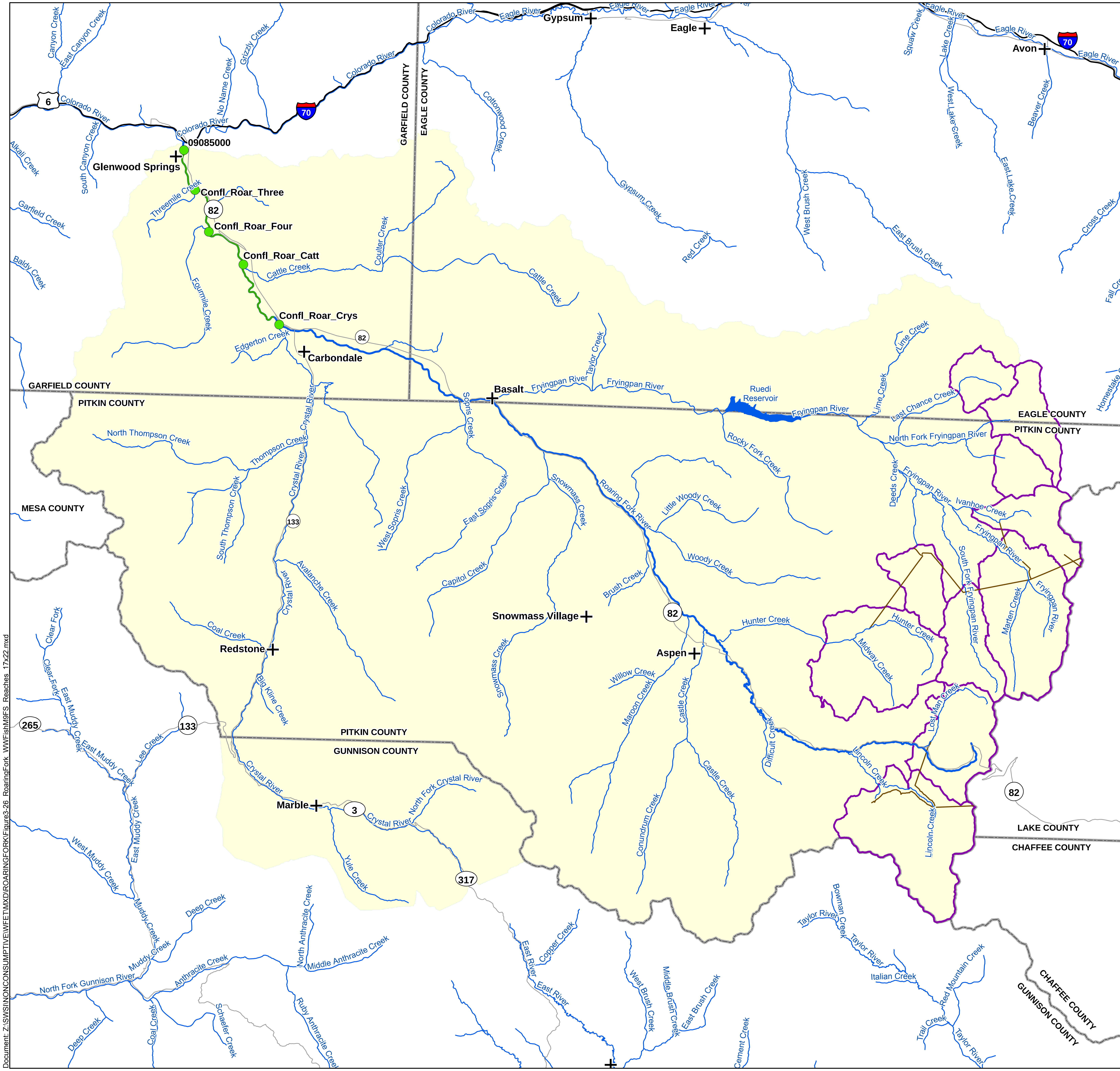

Legend

Warm Water Fish Method 9
Flannelmouth Sucker
Percent Reduction in Biomass
from Baseline to Existing Conditions

- < 10% - Low Ecological Risk
- 10 - 25% - Minimal Ecological Risk
- 26 - 50% - Moderate Ecological Risk
- ≥ 51% - High Ecological Risk
- < 10% - Low Ecological Risk
- 10 - 25% - Minimal Ecological Risk
- 26 - 50% - Moderate Ecological Risk
- > 50% - High Ecological Risk
- Transbasin Diversion Areas
- Lakes and Reservoirs
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

0 2 4 8
Miles
1 inch = 3.25 miles

Figure 3-26
Roaring Fork Watershed
Flow Evaluation Tool

Flannelmouth Sucker Risk
Reach Mapping

DRAFT

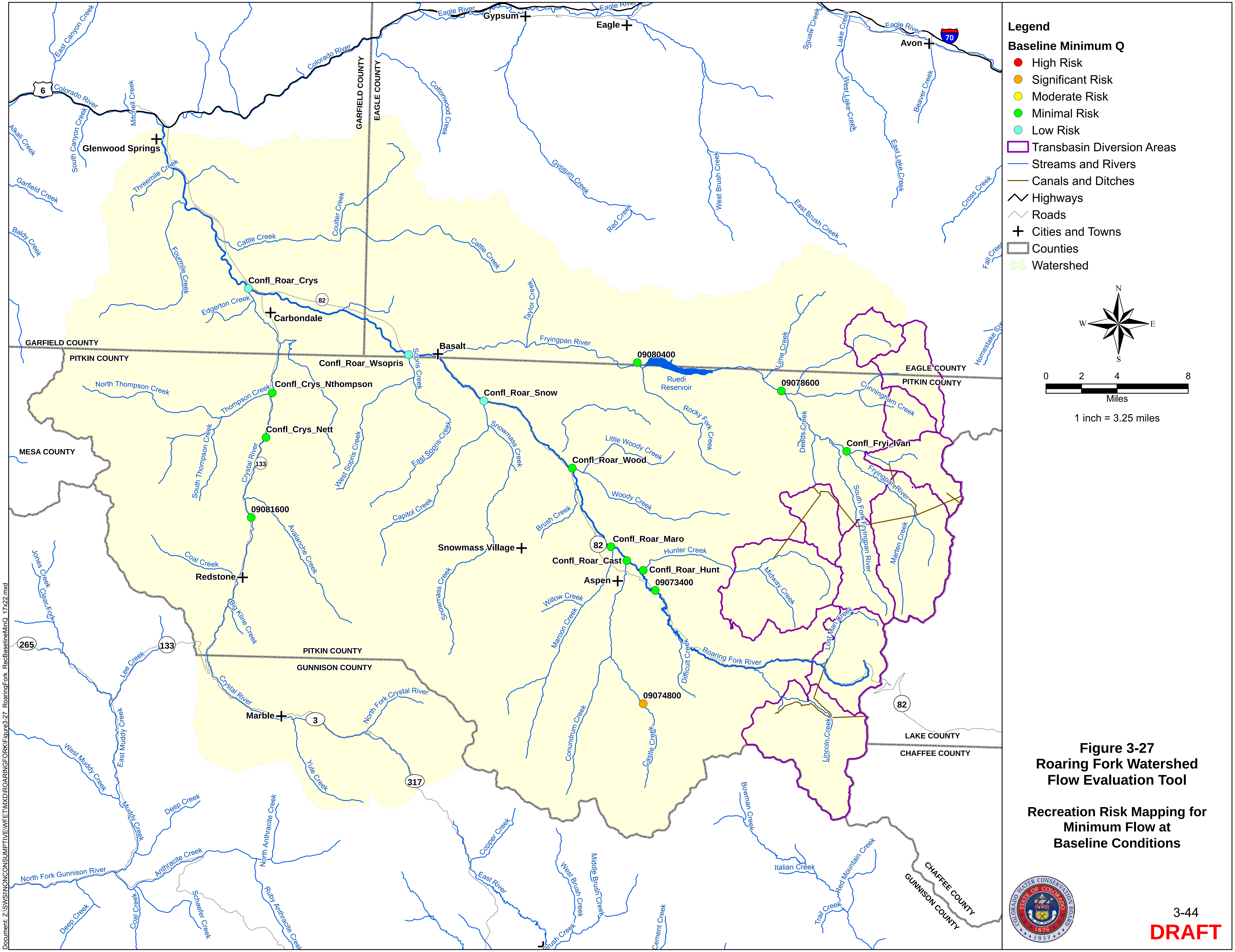

Figure 3-27
Roaring Fork Watershed
Flow Evaluation Tool

Recreation Risk Mapping for
Minimum Flow at
Baseline Conditions

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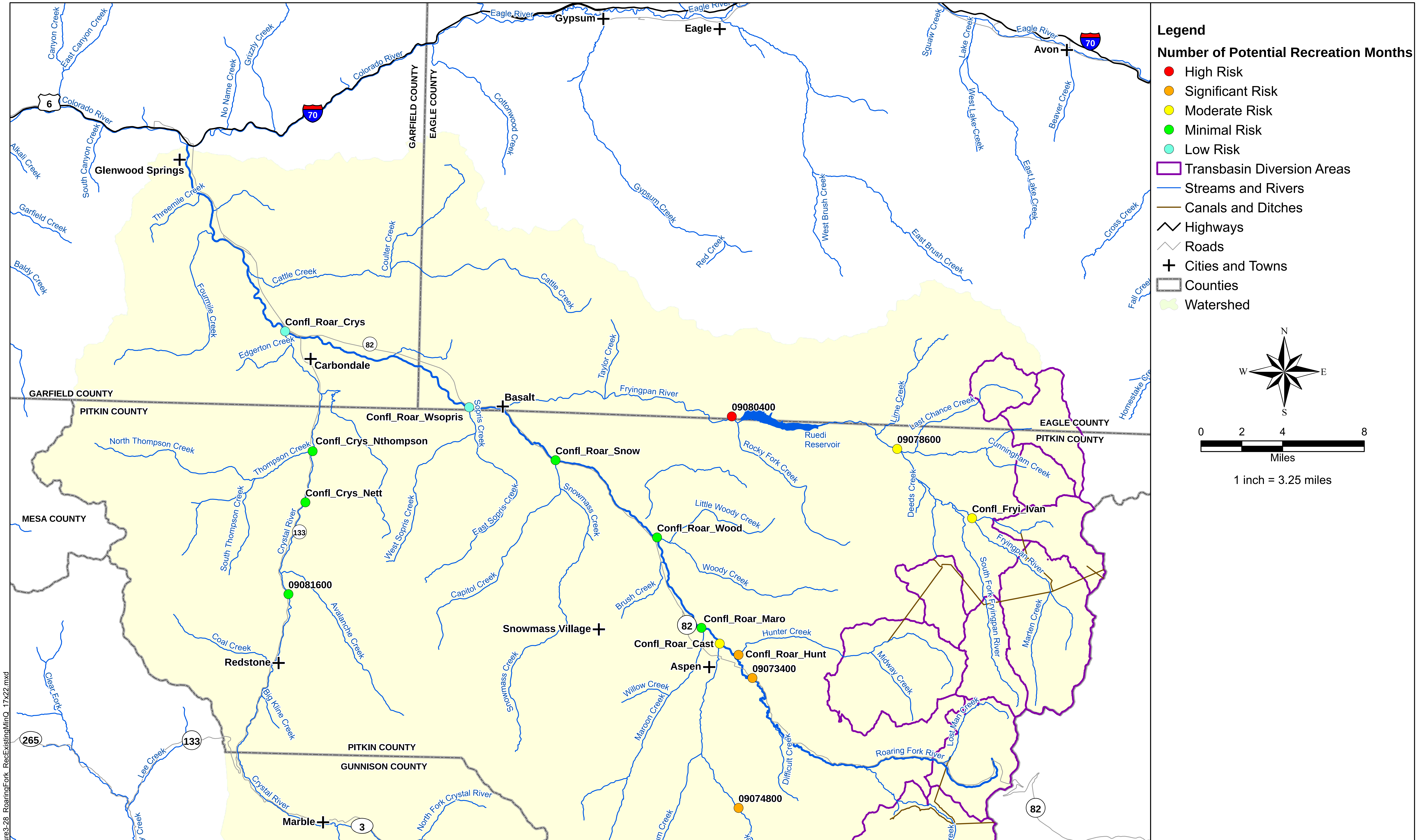


Figure 3-28
Roaring Fork Watershed
Flow Evaluation Tool

Recreation Risk Mapping for
Minimum Flow at
Existing Conditions

DRAFT

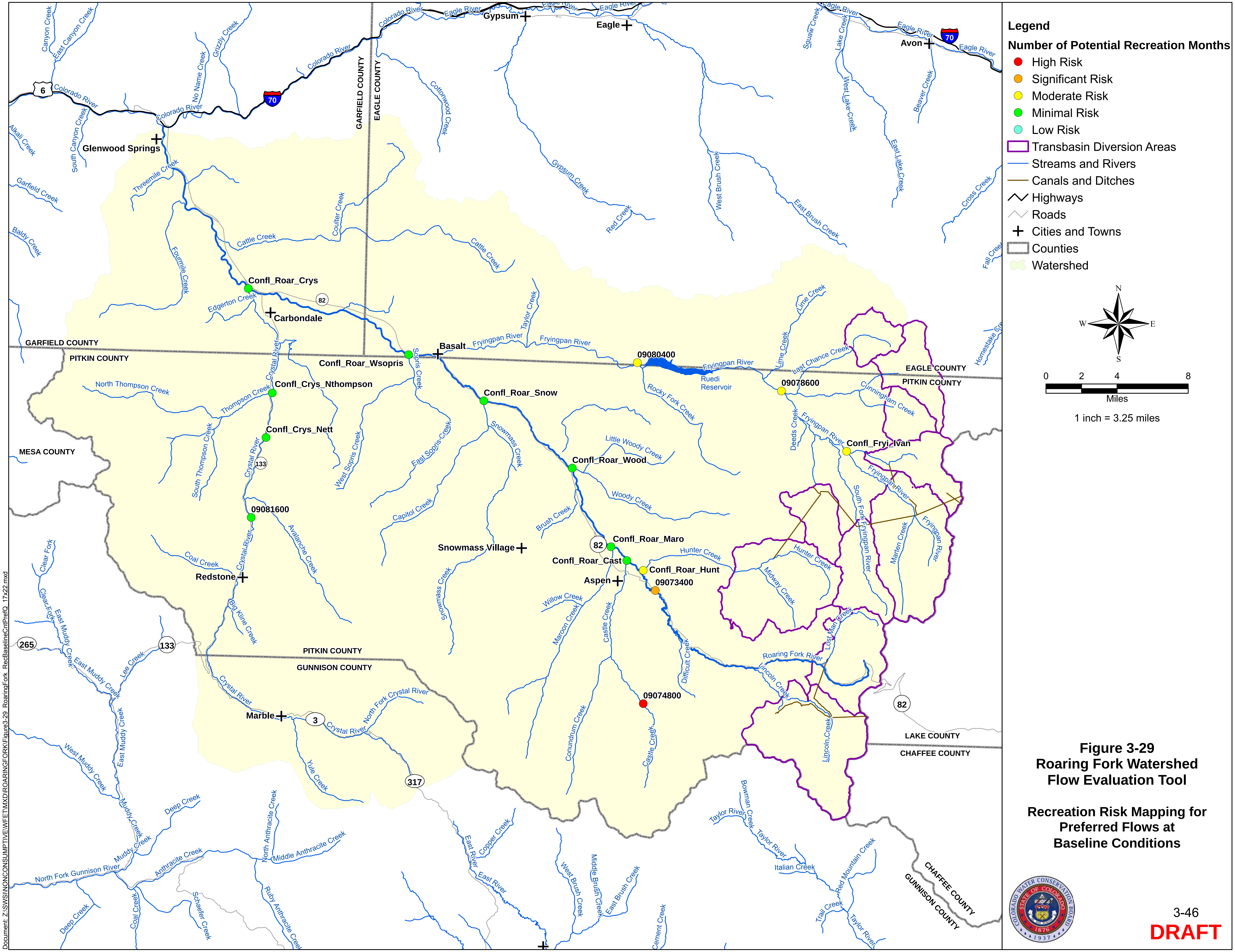
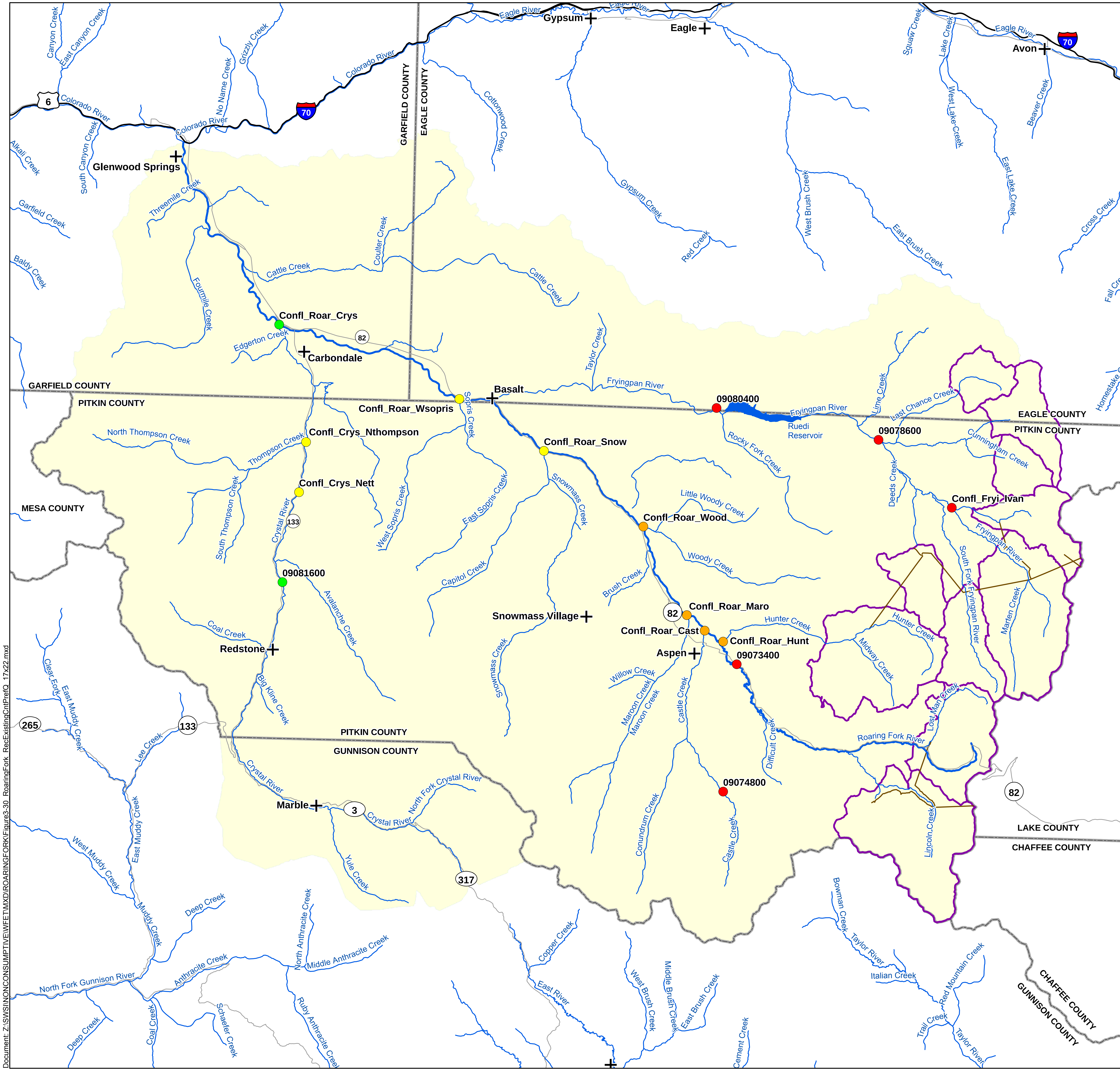


Figure 3-29
Roaring Fork Watershed
Flow Evaluation Tool

Recreation Risk Mapping for
Preferred Flows at
Baseline Conditions

DRAFT



Legend

Number of Potential Recreation Months

- High Risk
- Significant Risk
- Moderate Risk
- Minimal Risk
- Low Risk
- Transbasin Diversion Areas
- Streams and Rivers
- Canals and Ditches
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

N W E S

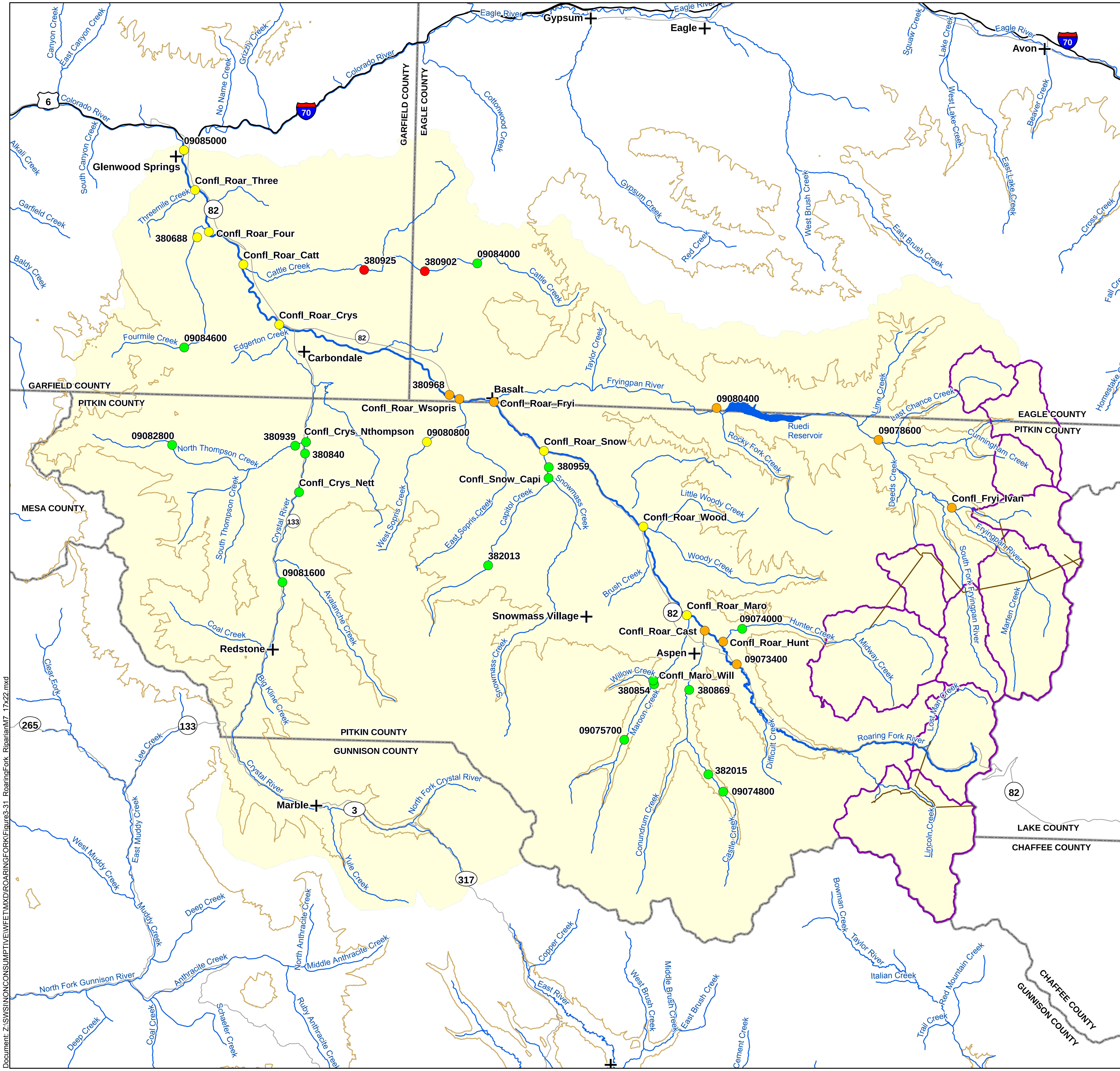
0 2 4 8
Miles

1 inch = 3.25 miles

Figure 3-30
Roaring Fork Watershed
Flow Evaluation Tool

Recreation Risk Mapping for
Preferred Flow at
Existing Conditions

DRAFT



Legend

Riparian Method 7
Change in Riparian Communities from Baseline to Existing Conditions

- < 10% - Low Risk
- 10 to 25% - Minimal Risk
- 26 to 50% - Moderate Risk
- > 50% - High Risk
- Transbasin Diversion Areas
- Lakes and Reservoirs
- Streams and Rivers
- Canals and Ditches
- 9600 Ft. Contour
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

N E S W

0 2 4 8
Miles

1 inch = 3.25 miles

Figure 3-31
Roaring Fork Watershed
Flow Evaluation Tool

Riparian Percent Change Risk Mapping

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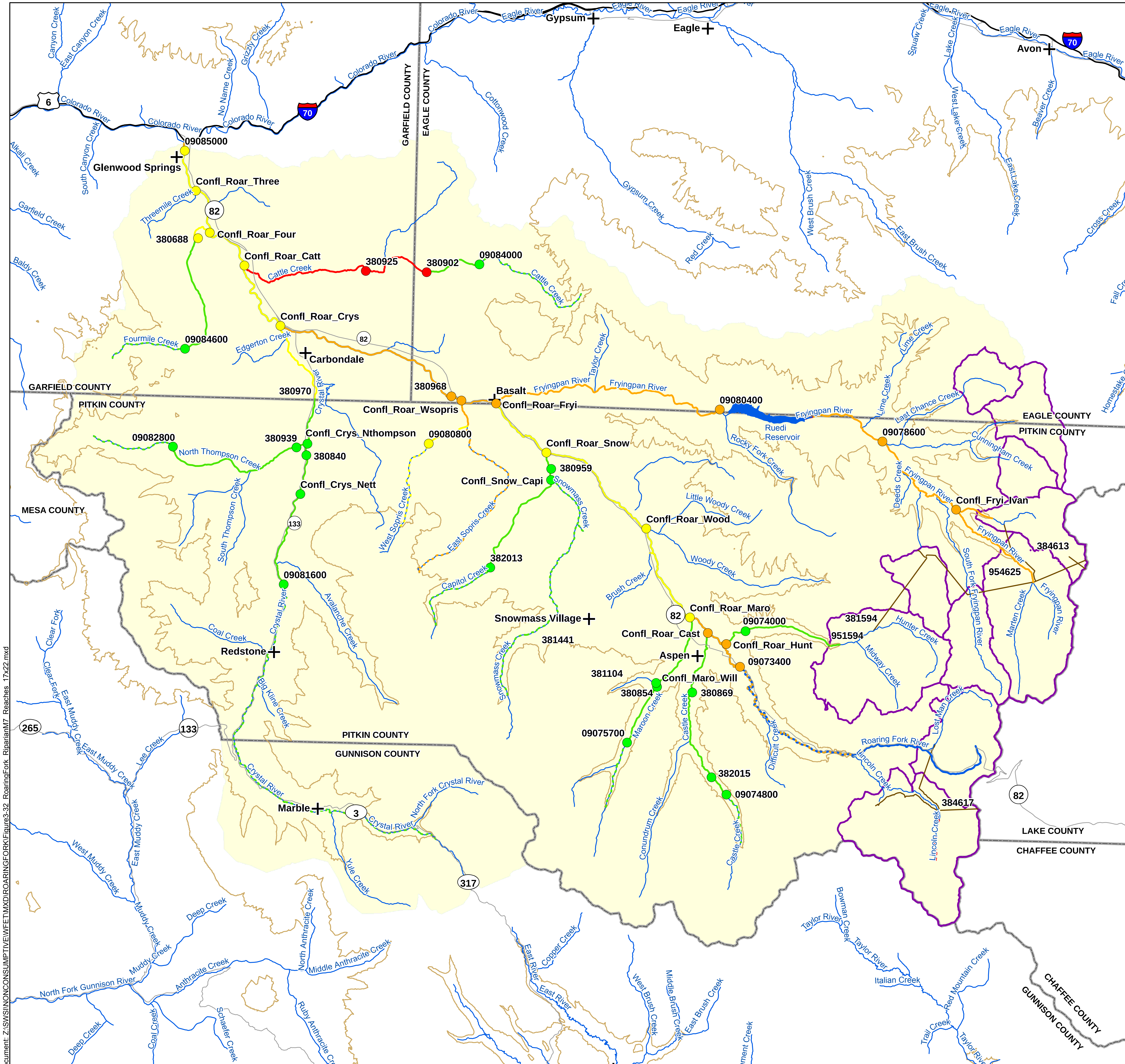

Legend

Riparian Method 7
Change in Riparian Communities from Baseline to Existing Conditions

- < 10% - Low Change
- 10 - 25%
- 26 - 50%
- > 50% - High Change
- < 10% - Low Risk
- 10 - 25% - Minimal Risk
- 26 - 50% - Moderate Risk
- > 50% - High Risk
- Transbasin Diversion Areas
- Lakes and Reservoirs
- Streams and Rivers
- Canals and Ditches
- 9600 Ft. Contour
- Highways
- Roads
- Cities and Towns
- Counties
- Watershed

0 2 4 8
Miles
1 inch = 3.25 miles

Figure 3-32
Roaring Fork Watershed
Flow Evaluation Tool

Riparian Percent
Change Risk Reach Mapping

DRAFT

Document: Z:\SWSI\NONCONSUMPTIVE\WFET\MXD\ROARINGFORK\Figure3-32_RoaringFork_RiparianM7_Reaches_17x22.mxd

Section 4
Watershed Flow Evaluation Tool Pilot Study Findings and Recommendations

4.1 Watershed Flow Evaluation Tool Pilot Study Findings

As discussed in Section 1, the results of the WFET pilot study for the Roaring Fork and Fountain Creek were considered in two ways. First, the results were examined based on whether the steps discussed in the research could be applied in Colorado in a manner that would provide meaningful technical results. Second, results were reviewed to provide recommendations for further application and refinement of the WFET.

4.1.1 Watershed Flow Evaluation Tool Technical Findings

The technical findings of applying the WFET in the Fountain Creek and Roaring Fork watersheds are as follows:

- Flow-ecology curves or relationships relating flow to ecological attributes were developed for key environmental and recreational attributes across the state of Colorado (Appendix B).

- Ecological risk mapping was generated in both the Fountain Creek and Roaring Fork watersheds for key environmental and recreational attributes.

- The majority of flow-ecology relationships are based on analyzing the ecological risk of removing water from a system. Thus, for Fountain Creek, where flows increased overall, the results regarding ecological risk for trout and warm water fishes are inconclusive.

- A more detailed analysis of Fountain Creek could be completed if a detailed hydrological model that could generate baseline and existing hydrologic conditions were available. This analysis would allow for better spatial understanding of hydrology throughout the system. Without such a model, review and input into the study results by local stakeholders were critical.

- Initial onsite validation on the Roaring Fork River between Basalt and Carbondale indicated that the WFET results are comparable with the site-specific results for trout.

- The recreation methodology needs more refinement in defining risk levels based on local knowledge and current site-specific studies that are being conducted as part of the Wild and Scenic process for the Colorado River.

4.1.2 Watershed Flow Evaluation Tool Application Findings

The findings regarding further application of the WFET in Colorado are as follows:

- The WFET provides a watershed scale, science-based perspective on ecological risks throughout drainage networks where site-specific studies are sparse or lacking.

- The WFET is best utilized in areas with detailed hydrologic data or models for pre and post water management conditions. The most logical continued use of the WFET in the near future is on the west slope were CWCB's DSS models are available.

- In areas where CWCB's DSS models are not available, the WFET could be used in a predictive capacity to examine potential future water management using conditions today as a baseline. For example, in the Arkansas and South Platte basins, one strategy for addressing future water needs is agricultural transfers. These transfers may alter the current flow regimes that support riparian and warm water fish attributes. The WFET could be used in a predictive capacity in these areas where today's flow regimes are considered baseline and where future flow regimes that could be modeled would be used to assess the ecological risk for current attributes.

- In Section 3.3 a potential application of the WFET for developing a range of seasonal flow conditions was presented, and further development of this approach could be used in regional water availability modeling such as the Colorado River Water Availability Study. Further development needed in applying this approach includes consideration of intra-year and inter-year hydrologic variability.

- The WFET could be used to help target instream flow acquisitions as well as restoration efforts in areas where it is applied in the future. For example, the WFET may identify areas where flow is not a limiting factor in an attributes ecological risk. This could indicate that an area is a good target for a restoration project. If portions of a watershed indicated ecological risk related to flow these areas may candidates for future instream flow acquisitions.

- The WFET is not intended to set flow prescriptions or rules for flow needs to the level of detail that would be required in a National Environmental Policy Act analysis or that might be needed to guide day-to-day management of a flow in a specific water project.

- The WFET could be used to build upon both the nonconsumptive needs assessment focus mapping completed by the Basin Roundtables and the work on strategies for Colorado's water supply future that CWCB is completing. The WFET could be used by the Basin Roundtables to examine which of their focus areas have attributes with ecological risk and could help focus the projects and methods to meet nonconsumptive needs in these areas. With regard to strategy development, the

WFET could be used to identify if there are areas at risk where a strategy is being considered for further development.

- The WFET and site-specific studies are complementary and interrelated. The WFET could be used to target site-specific studies in critical locations. The targeted site-specific studies could in turn be used to refine ecological risk categories for subsequent WFET mapping across different geographies.

4.2 Watershed Flow Evaluation Tool Pilot Study Recommendations

Following are recommendations for further improvement of the WFET if it is applied in the future in Colorado:

- Further validation of the WFET should be completed. This should be based on other site-specific studies, particularly CWCB instream flow R2CROSS data. In addition to trout and warm water fishes site-specific studies, site-specific studies related to riparian areas should also be considered in further validation efforts.

- The ranges of ecological risk are based on literature and should be further refined with site-specific data. Further calibration of ecological risk levels with site-specific data should occur. It is important to note that site-specific data used for calibration of risk levels would not be used for WFET validation.

- The ranges of risk for recreation should be further refined with site-specific data.

- The WFET analyzes ecological risk mapping between nodes. Further refinement of ecological risk between and above nodes should be completed in future efforts. In some areas, conditions in the headwaters could be further refined based on knowledge of diversions or consumptive use patterns.

- The WFET application of developing seasonal flow conditions based on the ecological risk levels will require refinement considering intra-year and year-to-year variability.

- Application for input into the Colorado River Water Availability Study needs to be further developed as described above, and a method needs to be developed for determining winter baseflow levels where there is not an instream flow.

- Calibration and validation processes need to be refined. Calibration based on site-specific studies should be used to adjust risk levels for all metrics.

Section 5
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Appendix A

Nonconsumptive Needs Assessment: Roaring Fork River Site-Specific Pilot Study

Draft Report

Non-Consumptive Needs Assessment: Roaring Fork River Site Specific Pilot Study

Prepared for:
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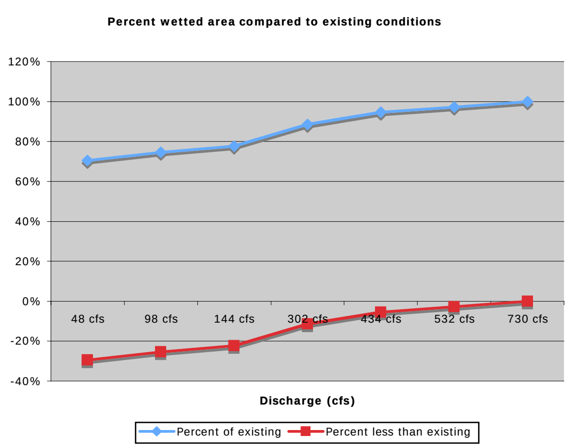
Denver, Colorado 80202



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Revised June 22, 2009

MILLER ECOLOGICAL CONSULTANTS, INC.

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Introduction

As part of the Interbasin Compact Process, each Basin Roundtable is developing a non-consumptive needs assessment. The goals of the process include identifying priority areas and stream reaches for environmental and recreational attributes, and identifying quantities of seasonal flows needed to maintain those area and reaches. There are two approaches under consideration to develop the quantifications, first a coarse scale method that would evaluate large watershed areas, and second, a fine scale method that would evaluate a specific river reach. The latter method is the focus of this report. The Colorado River Basin Roundtable Non-Consumptive Needs subcommittee decided to conduct a pilot study of the site specific quantification. The river reach chosen was the Roaring Fork River, which had an existing data set that could be used for the site specific quantification. In addition, a Watershed Flow Evaluation Tool (WFET) developed for the statewide process was being tested on the Roaring Fork River basin.

There are two main objectives for the pilot study, 1) use the site specific approach to evaluate the change between baseline and existing conditions, and 2) evaluate high risk conditions identified with the WFET using the site specific approach. The second objective also allows an evaluation of how the coarse and fine scale evaluations work in conjunction with each other.

Study Area

The site specific evaluation relied on existing data for the Roaring Fork River that was collected downstream from the Fryingpan River. The data is representative of the river reach between the Fryingpan River and the Crystal River (Figure 1). The Roaring Fork River from the confluence of the Fryingpan River downstream through the town of Carbondale consists of a fairly uniform mixture of riffle/run habitats (Figure 2). Pool habitat is relatively uncommon throughout this reach. This channel type is referenced as

Roaring Fork River Tree Farm (RFR-TF). This representative site was established on the "Tree Farm" USFS land (Figure 1).

Figure 1. Study area for Roaring Fork River Site Specific Pilot Study.

Figure 2. Site RFR-TF on the Roaring Fork River, June 2001, at 876 ft^3/s. Reported discharge is from USGS gaging station 09081000.

Methods

The data used in the Roaring Fork River Site Specific Pilot Study was collected during the Fryingpan Roaring Fork Fishery Study (Ptacek et al. 2003). That study included an application of the Physical Habitat Simulation System (PHABSIM) at one site on the Roaring Fork River downstream from the Fryingpan River. The methods from that study are described below.

Data Collection

The following methodology applies to specific techniques applied to the Roaring Fork River.

Transect placement followed the criteria proposed by Bovee (1982) and Bovee (1997). Transects were placed (marked with wooden stakes) in all habitats that represented over five percent of the total available habitat. The number of transects placed in each habitat type depended on the physical and hydraulic features of each location. Transects were placed in homogeneous habitat types. Additional transects were placed at key hydraulic locations within the habitat type to ensure better model calibration and simulation. Transects were located in contiguous habitats.

Data required by IFIM includes a full set of hydraulic measurements (bed and velocity profiles, water surface elevations, and discharge) and several stage-discharge measurements. Vertical elevations were established throughout each habitat type by establishing a primary benchmark and at least two secondary benchmarks at each study site. At each habitat and hydraulic transect, a measuring tape was stretched across the river and attached to the wooden stake representing the end of that specific transect. Linear distance (stationing) between stakes was recorded for all measured parameters. Streambank and water surface elevations were surveyed using a standard auto level and differential leveling. All surveys followed general guidelines listed by Bovee (1997). Within the stream channel, depth and mean column velocity were measured every 1-3 ft. across the wetted portion of the river. A Swoffer Model 2100 velocity meter and topset rod were used for all discharge and velocity profile measurements. Along the transect line at each interval where depth and mean column velocity were measured, dominant and subdominant substrate (following codes from Bovee (1997)) and cover type were also recorded.

<u>Target Species:</u>

The PHABSIM analysis used adult and juvenile life stages of rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*) as the target species. Adult and juvenile habitat suitability curves were developed by CDOW and USGS on the South Platte River below Cheesman Dam, near Deckers, Colorado. The South Platte River in this area is a tailwater stream with a large salmonid population composed of naturally reproducing brown and rainbow trout.

Hydraulic Simulations

All field data were entered into a spreadsheet program and checked for accuracy. The windows based PHABSIM version 1.10 software (USGS Mid-continent Ecological Science Center 2001) was used to create the hydraulic modeling runs. PHABSIM combines hydraulic modeling programs with a habitat suitability subroutine, allowing the user to predict changes in physical habitat due to alterations in flow.

In addition to the field data collected, PHABSIM requires the input of reach slope and habitat weighting factors. Slope was calculated for each reach using the water surface elevations and distance from the most upstream transect to the most downstream transect. Therefore, reach length and habitat weighting factors were determined using the "habitat typing" technique, which is the preferred technique (Bovee 1989).

Each site was calibrated to measured water surface elevations and velocity distributions. Water surface elevations and velocities were modeled for simulated flows using the calibration corrections. Specific flows simulated ranged from 48 ft^3/s to 2000 ft^3/s on the Roaring Fork River. The computer programs Avparm and Avdepth (submodels of PHABSIM) were run to determine wetted perimeter, average depth and average velocity for each cross section at each simulated flow.

Using the PHABSIM submodel HABTAE, habitat suitability curves were run to determine weighted usable area (WUA) for rainbow trout and brown trout juveniles and adults. Weighted usable area values are reported as $feet^2$ per 1,000 feet of river to allow direct comparison between modeled sites.

Several analysis techniques were used to interpret the PHABSIM output. Habitat time series (Bovee 1982), WUA versus discharge (Bovee 1982), and wetted perimeter

(Wesche and Rechard 1980; Leathe and Nelson 1986) techniques were used to analyze the effect of flow regime modification on trout habitat.

The wetted perimeter technique evaluates the decline in wetted perimeter as a function of discharge. Based upon this relationship, an “inflection” point was determined for riffle transects. Below the inflection point threshold, wetted perimeter declines rapidly for relatively small reductions in discharge (Annear and Condor 1984). The inflection point method provides another tool in the process of analyzing the affects of particular flow regimes on the aquatic communities.

Habitat time series analysis allows the direct comparison of multiple flow regimes on the trout habitat quality. The hydrology for the habitat time series was developed from the STATEMOD hydrology model. The analysis was comprised of the existing flow regime and the baseline conditions for the undiverted flows.

RESULTS

IFIM Hydraulic Modeling

During 2001 at total of four discharges were measured on the Roaring Fork River (Table 1). A total of 5 transects were established at the Roaring Fork River site (Table 2).

Generally, there is more rainbow trout habitat than brown trout habitat over all flows simulated. In addition, juvenile habitat was low compared to adult habitat, averaging less than 30 percent of adult habitat maximum WUA values (Figure 3 , Figure 4). The IFIM site on the Roaring Fork River (RFR-TF) had a higher optimum habitat for rainbow trout habitat than brown trout habitat. Most maximum WUA values occurred at approximately 300 ft^3/s.

Table 1. IFIM measurements conducted during 2001 on the Roaring Fork River.

Date	Discharge (ft^3/s)	Measurement
22 June 2001	876	Water surface elevations, Habitat mapping
9 July 2001	571	Water surface elevations
31 July 2001	379	Water surface elevations
11 October 2001	302	Bed profiles, Water surface elevations

Note: Reported discharges are from USGS gaging station 09081000.

Table 2. Instream Flow Incremental Methodology (IFIM) transect designations for sites on the Roaring Fork River, Colorado.

Site	Transect	Habitat Type
RFR-TF	1	Riffle
	2	Run control
	3	Run
	4	Riffle control
	5	Run

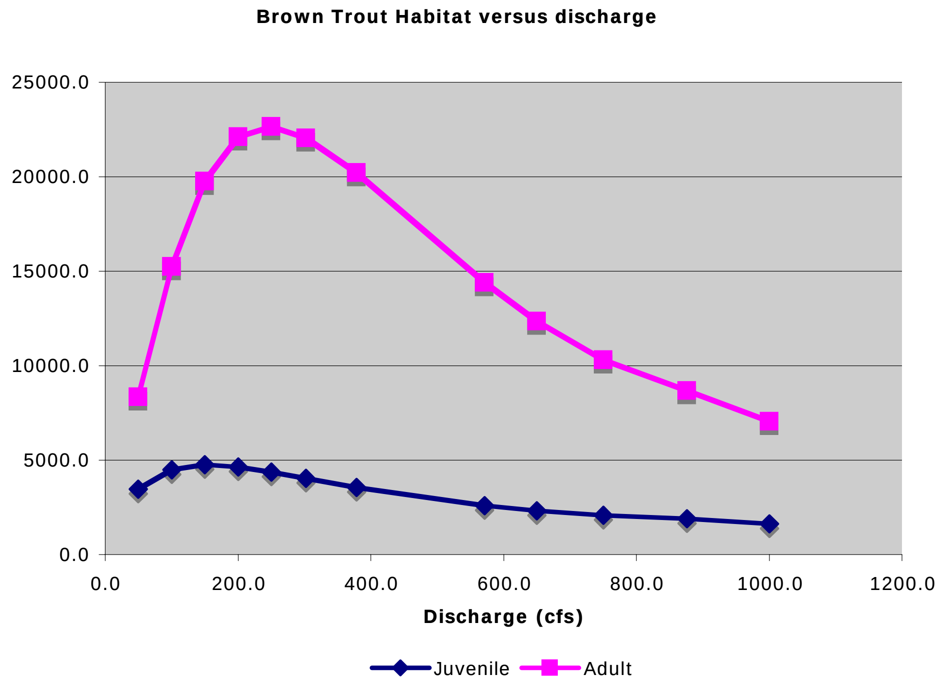

Figure 3. Weighted usable area (ft^2 per 1,000 ft) for brown trout versus discharge (ft^3/s) for RFR-TF.

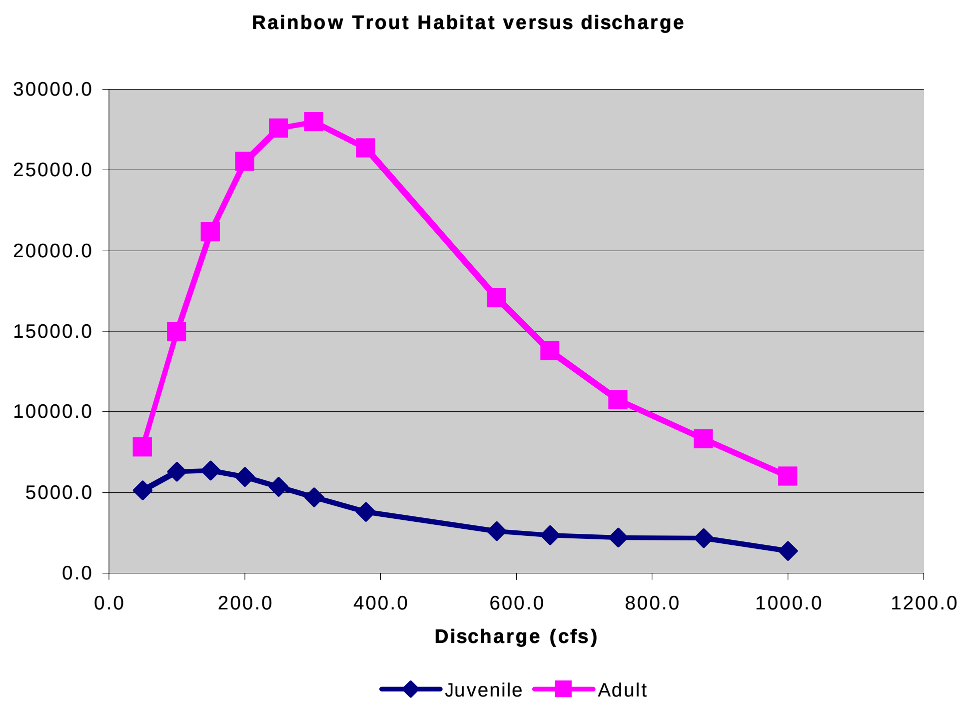

Figure 4. Weighted usable area (ft^2 per 1,000 ft) for rainbow trout versus discharge (ft^3/s) for RFR-TF.

Comparison of Existing and Baseline habitat conditions

Existing flows can range from more than 2000 cfs at peak to less than 300 cfs during summer. The stream channel is wet from bank to bank at flows higher than 700 cfs (Figure 5). When flows are near base flow condition, relatively large areas of cobble bars are exposed (Figure 6). The amount of stream channel that is wet or dry depends on the shape of the stream cross section. Wider more uniform cross sections have less change in wetted area than narrower, highly varied cross sections (Figure 7, Figure 8).

There is little difference in wetted area or water depth between the existing conditions and baseline conditions in August and September, regardless of channel type (Figure 9, Figure 10, Figure 11, Figure 12). Both channel types are wet from bank to bank at the baseline and existing flows. The fully wetted channel provides good conditions for fish and invertebrates. Since the habitat conditions are similar between baseline and existing flows, the trout habitat should be similar. This river reach has very good trout populations under existing conditions.

Figure 5. Site RFR-TF on the Roaring Fork River, June 2001, at 876 ft^3/s. Reported discharge is from USGS gaging station 09081000.

Figure 6. Site RFR-TF on the Roaring Fork River, October 2001, at 302 ft^3/s. Reported discharge is from USGS gaging station 09081000.

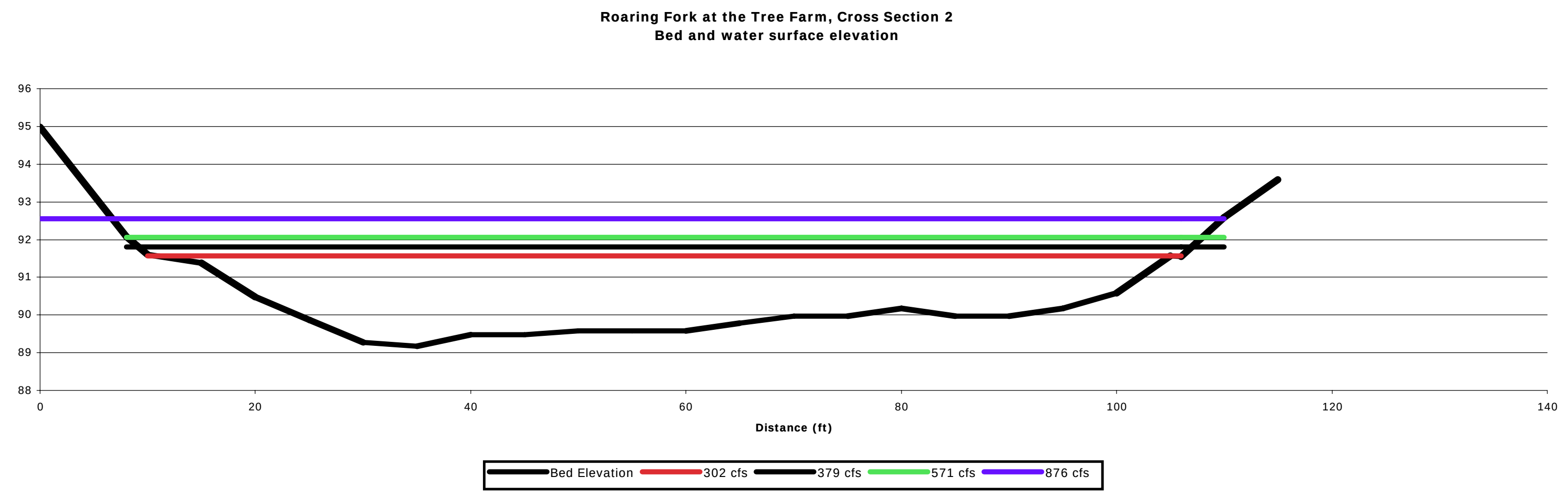

Figure 7. Water surface elevations measured at four discharges at Roaring Fork River Cross Section 2.

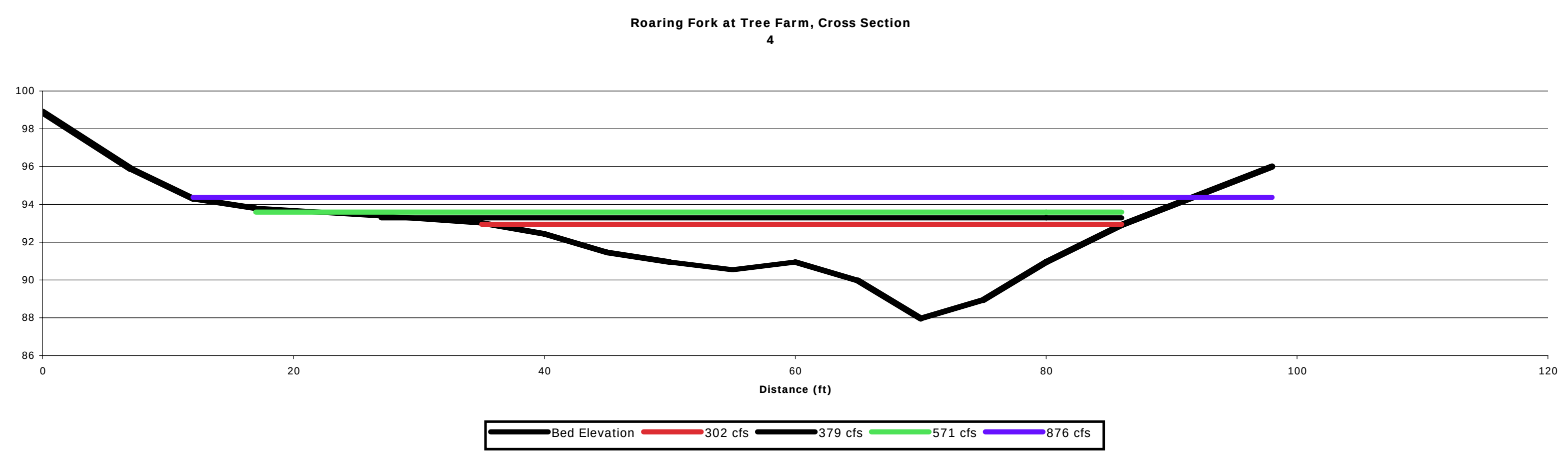

Figure 8. Water surface elevations measured at four discharges at Roaring Fork River Cross Section 4.

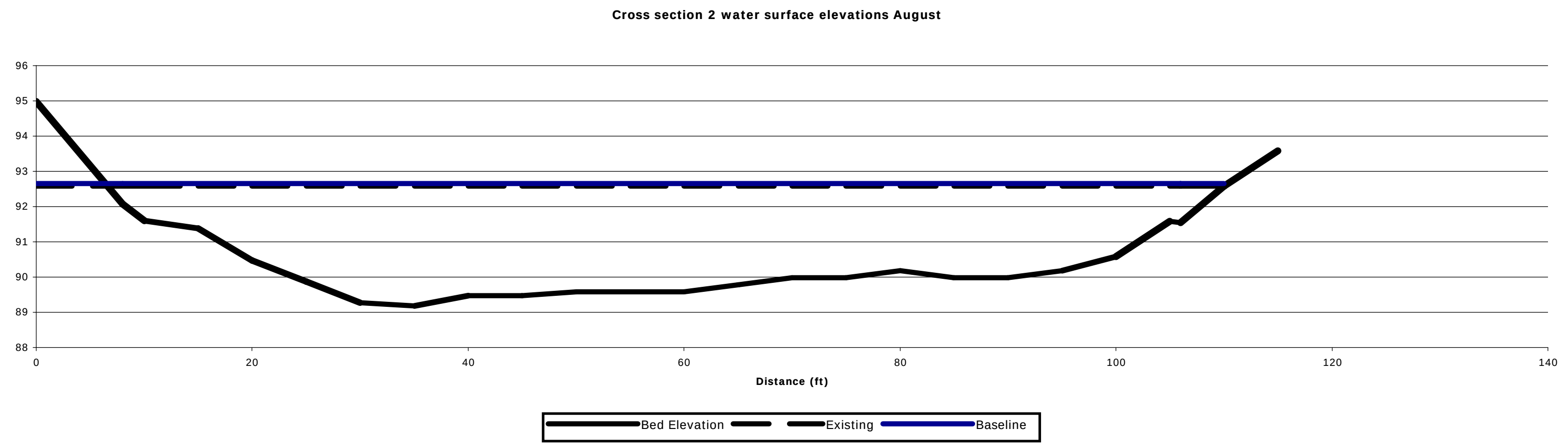

Figure 9. Comparison of baseline and existing water surface elevation for Roaring Fork River Cross Section 2 for average August discharge.

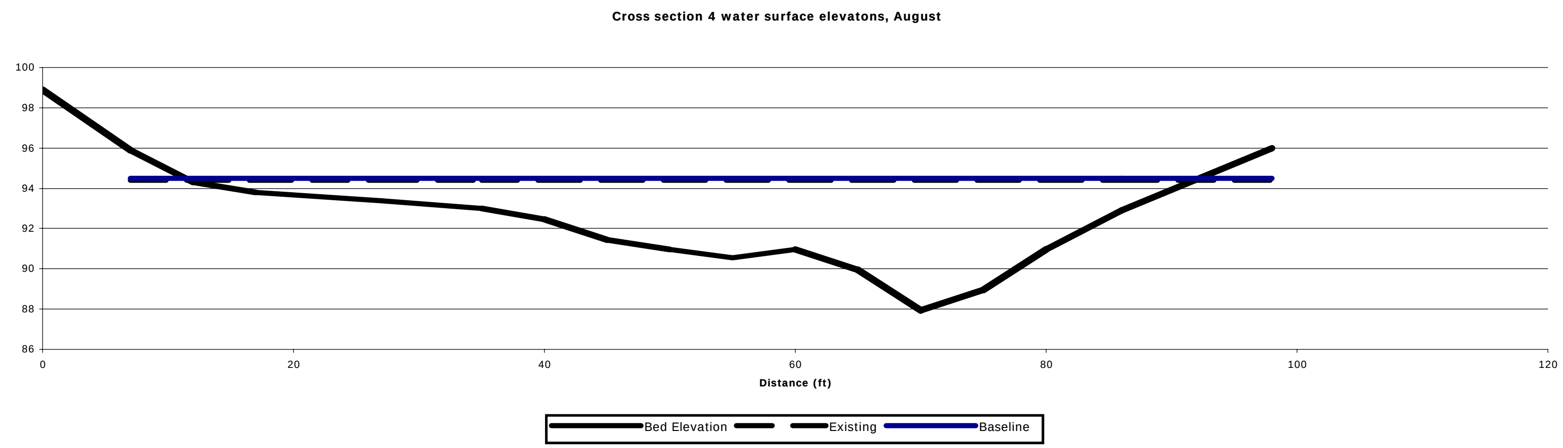

Figure 10. Comparison of baseline and existing water surface elevation for Roaring Fork River Cross Section 4 for average August discharge.

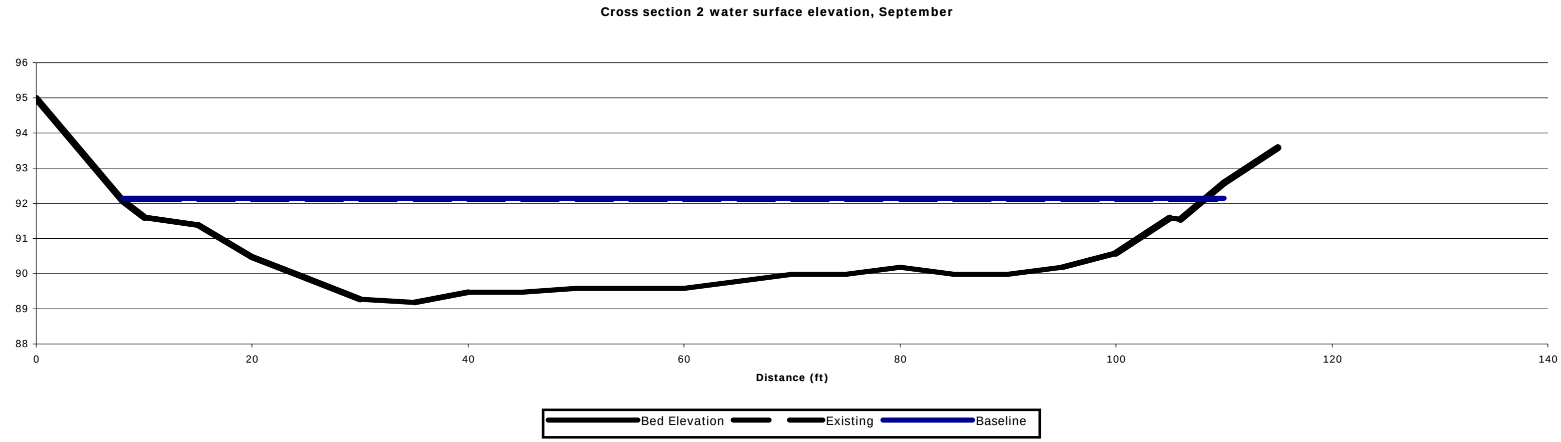

Figure 11. Comparison of baseline and existing water surface elevation for Roaring Fork River Cross Section 2 for average September discharge.

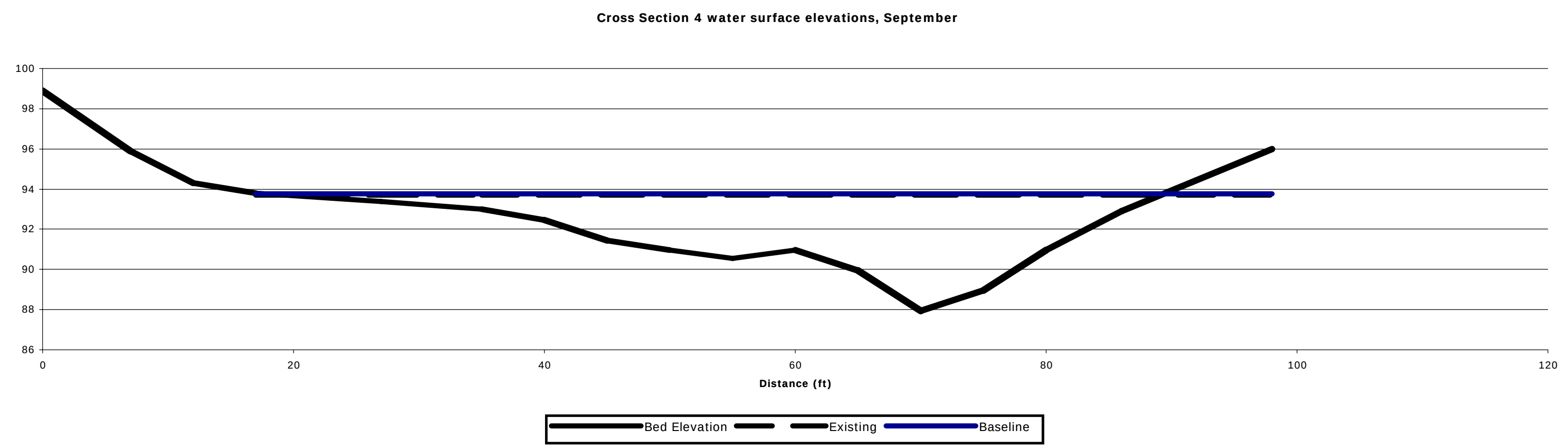

Figure 12. Comparison of baseline and existing water surface elevation for Roaring Fork River Cross Section 4 for average September discharge.

Comparison between existing conditions and high risk conditions predicted by the WFET.

The following discussion provides an example of the type of comparison that can be made between the WFET and the Site-Specific tools. The PHABSIM model can calculate multiple types of metrics related to hydraulics and habitat. The example presented here uses adult rainbow trout but the model produces the same metrics for any species or life stage that is simulated.

The WFET was used to determine flows that should be a high risk for trout populations. These flows were 48 cfs, 98 cfs, and 148 cfs, which correspond to poor, marginal and fair habitat based on the WFET metrics. An evaluation of those flows was made using the site specific data. Total wetted area as a function of discharge shows that wetted area at all three flows is substantially lower than the existing August-September average flow of 730 cfs (Figure 13). This is the result of the channel shape and the stage discharge function. There is a very large change in water surface with small changes in flows up to the point where the majority of the channel is wet (Figure 14). This relationship is also shown with the comparison of wetted area. There is a substantial difference between the percent of the channel that is wet at the high risk flow than at the existing flow (Figure 15). The comparison of the water surface elevations for the high risk and existing flows shows that this relationship holds for most of the individual cross sections (Figure 16 - Figure 20).

The habitat versus discharge function doesn't show this same relationship. There is approximately the same total amount of habitat for flows less than 100 cfs and 730 cfs (Figure 21). The maximum habitat for adult trout occurs at approximately 300 cfs. While the habitat quantity may be similar at the lower and higher flows, the habitat quality is different. PHABSIM calculates the weighted usable area by multiplying the area represented by a particular location by the combined suitability for depth, velocity, and substrate. This results in habitat quantification for the site that varies by location. In the comparison of 48 cfs with 730 cfs, the habitat at 48 cfs is lower quality and less river channel is available than at 730 cfs (Figure 22, **Error! Reference source not found.**). There is approximately 10 percent difference in total usable habitat for rainbow trout adult between 48 cfs and 730 cfs. However, the habitat at 730 cfs is of higher quality (see lower legend figures 22 and 23 for habitat quality). The legend shows the habitat quality by cross section. The habitat quality can be depicted in a three dimensional view as well to display channel shape in combination with habitat quality (Figure 24, Figure 25). There is approximately 10 percent difference in total usable area between 48 cfs and 730 cfs for adult rainbow trout, however, there is a substantial difference in habitat quality and wetted channel area. There is more habitat of higher quality at 730 cfs than at 48 cfs and there is more wetted channel area.

The above example show that the site specific approach can provide a robust characterization of habitat between two different flows, and an appropriate approach should compare total wetted channel, change in water surface, habitat quantity and habitat quality. The PHABSIM model has built in graphics for all of these metrics, however, the graphics are preset for the displays, axis titles and legends. The units for all the PHABSIM data are English (feet or square feet).

The most appropriate interpretation of the PHABSIM data should include a comparison of the multiple metrics that can be derived from the PHABSIM model including wetted area, wetted perimeter, and weighted usable area (quality and quantity) at multiple flows. These multiple hydraulic and habitat metrics should be included in the validation and calibration of future WFET applications.

Conclusions

The Site Specific approach quantifies changes for specific river reaches, species and river discharges. An application of the Site Specific approach requires existing data or collection of new data. Existing data can range from a single cross section to a detailed two dimensional hydraulic model. The use of a hydraulic model permits calculation of multiple metrics (e.g. water surface changes, habitat quantity changes, habitat quality changes) that are specific to the reach being studied. As such, this can provide a relatively large amount of detail for a selected reach of stream.

The additional detail allows for comparison of channel metrics (e.g. water depth, water width) between baseline and existing conditions for August and September flows. The comparison for the Roaring Fork shows very little change between existing and baseline conditions.

The existing high flows extend from bank to bank and help maintain the riparian community. The Analysis of habitat over time allows comparison of multiple flow regimes and evaluation of alternative flow management scenarios or different levels of ecological risk. The comparison of PHABSIM results with WFET high risk flows show that those flows are producing conditions that could result in degraded aquatic conditions. In general, site specific results validate the WFET results.

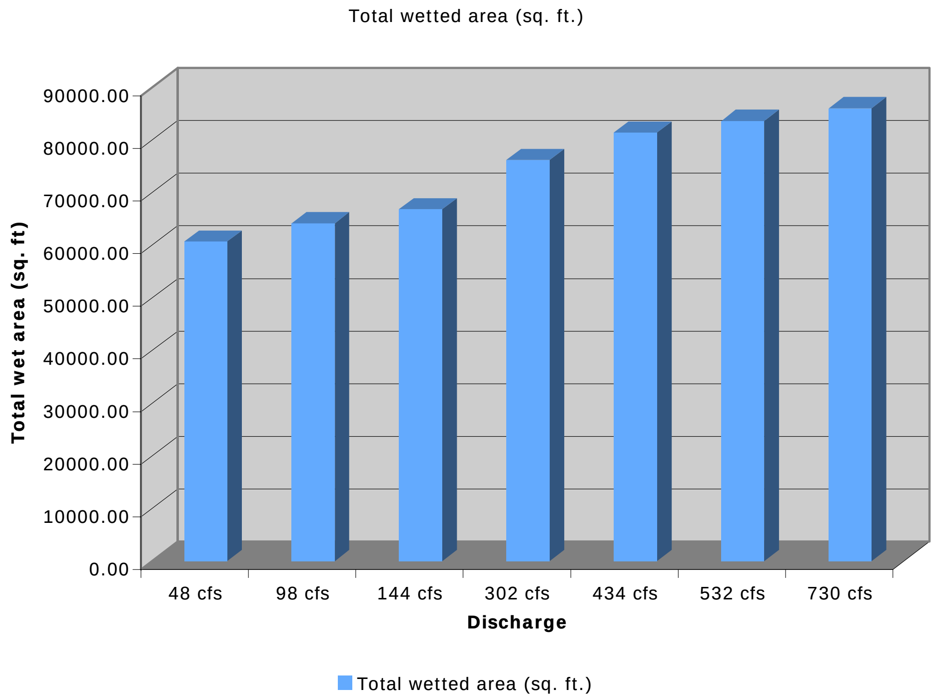

Figure 13. Total wetted area as a function of discharge for Roaring Fork River site RFR-TF.

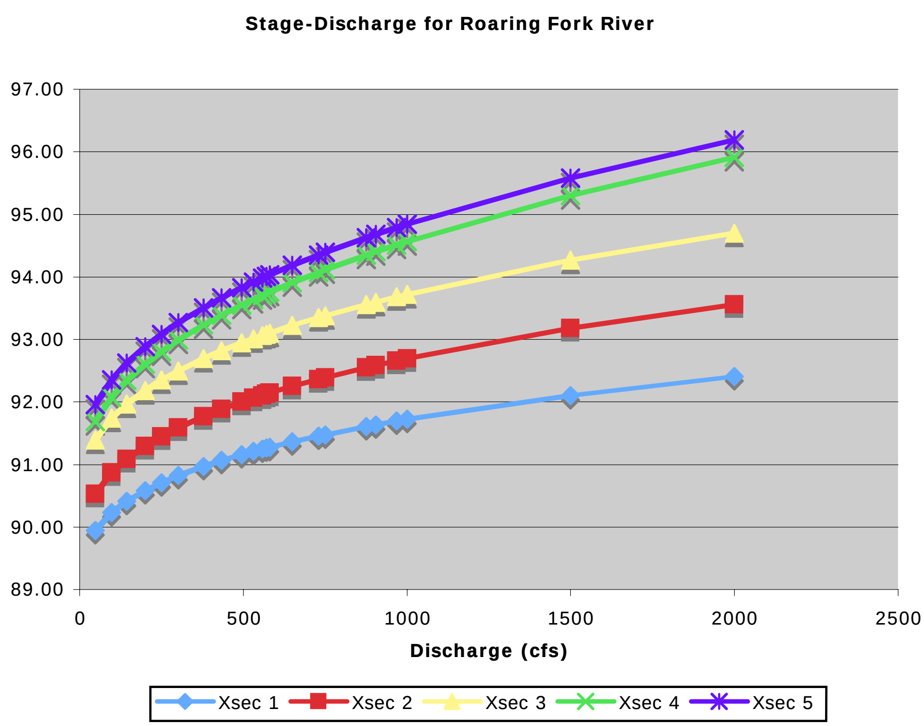

Figure 14. Stage discharge function for Site RFR-TF.

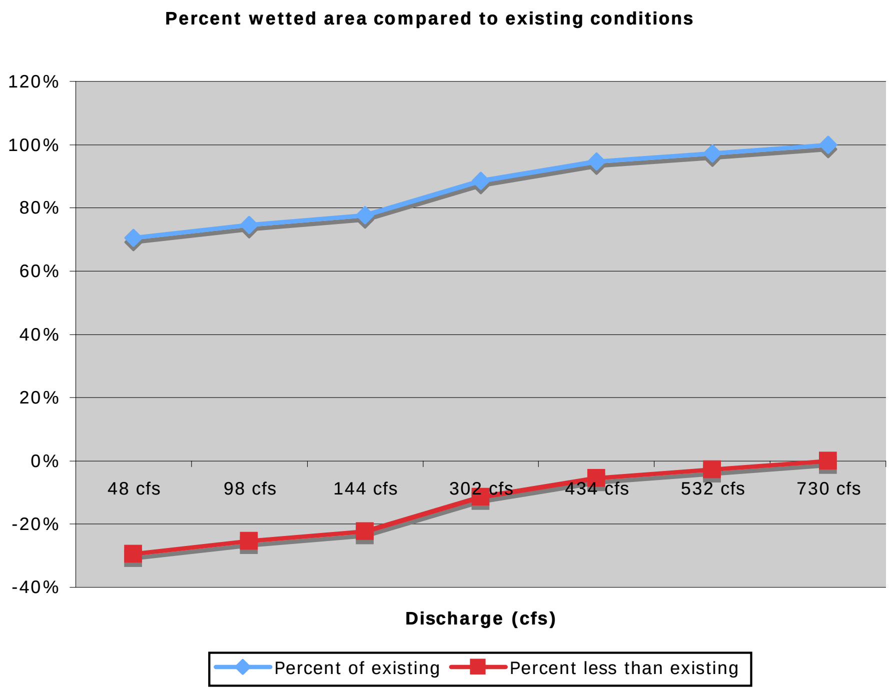

Figure 15. Percent wetted river channel at high risk flows compared to existing conditions at Site RFR-TF.

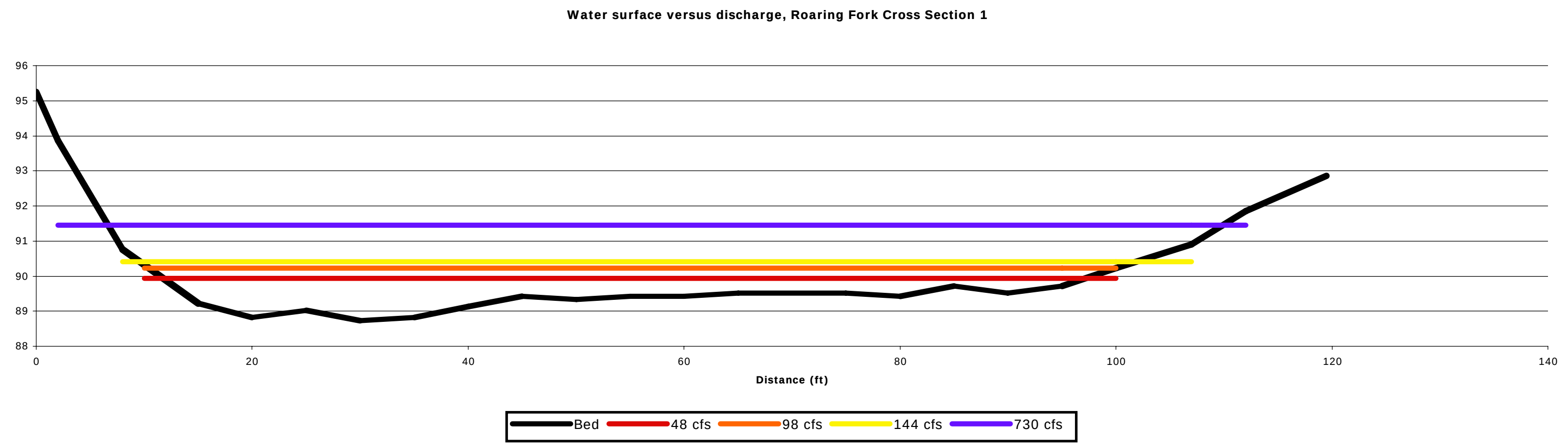

Figure 16. Comparison of water surface elevations for high risk flows and average August-September flows for RFR-TF Cross Section 1.

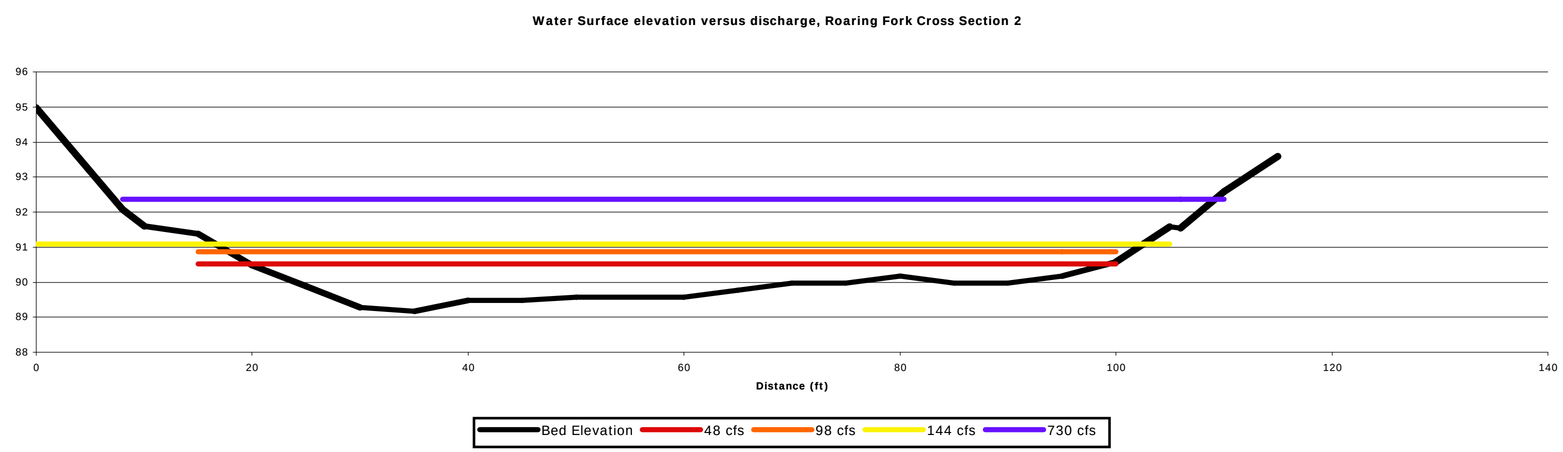

Figure 17. Comparison of water surface elevations for high risk flows and average August-September flows for RFR-TF Cross Section 2.

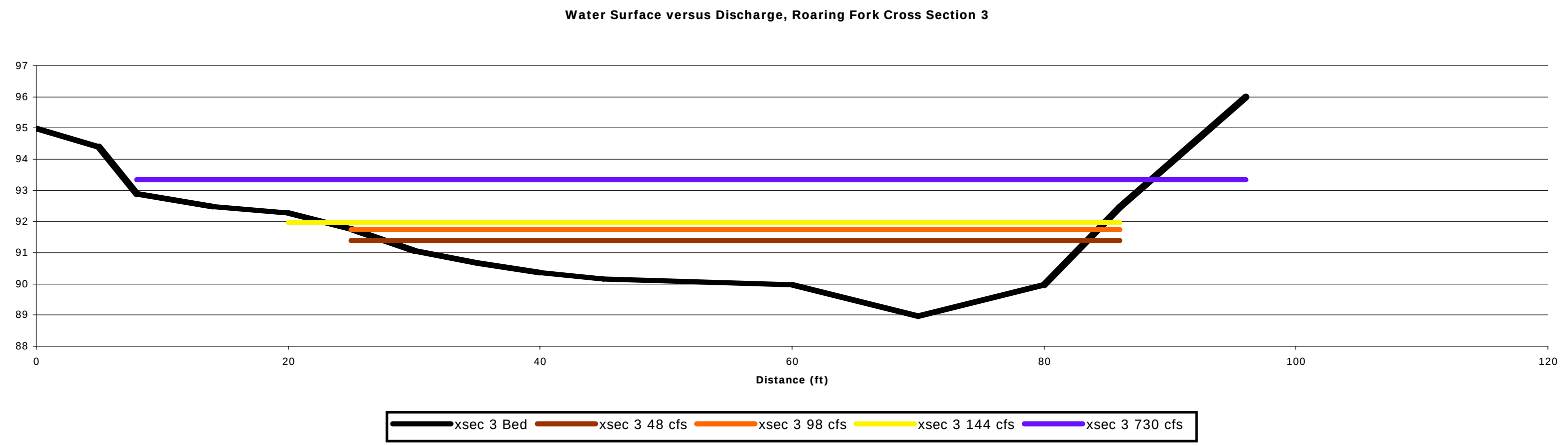

Figure 18. Comparison of water surface elevations for high risk flows and average August-September flows for RFR-TF Cross Section 3.

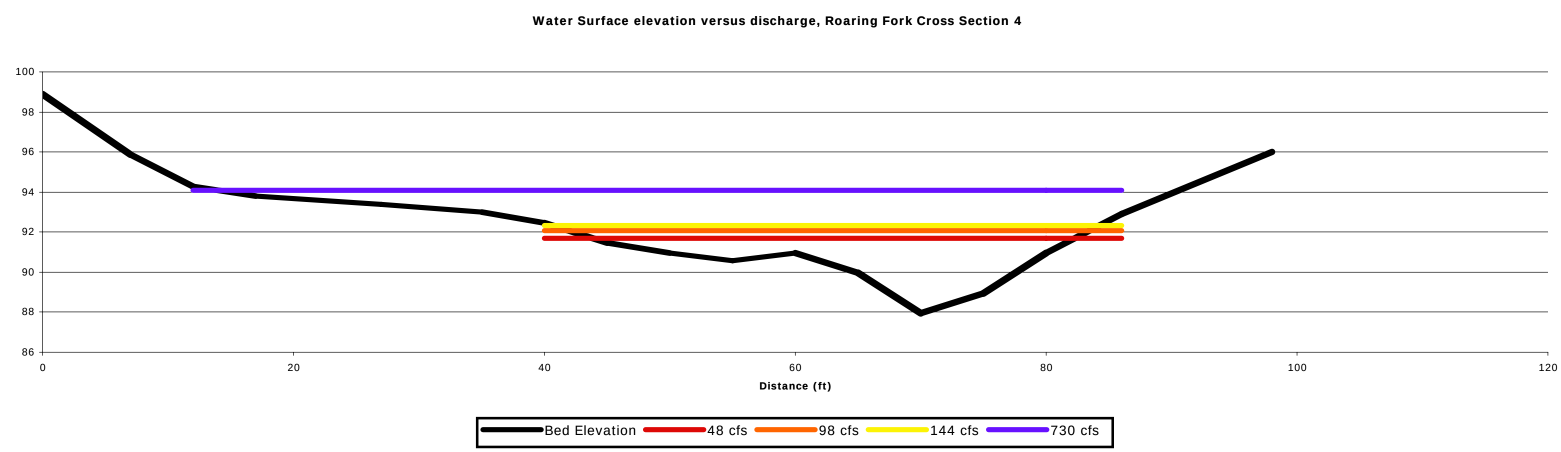

Figure 19. Comparison of water surface elevations for high risk flows and average August-September flows for RFR-TF Cross Section 4.

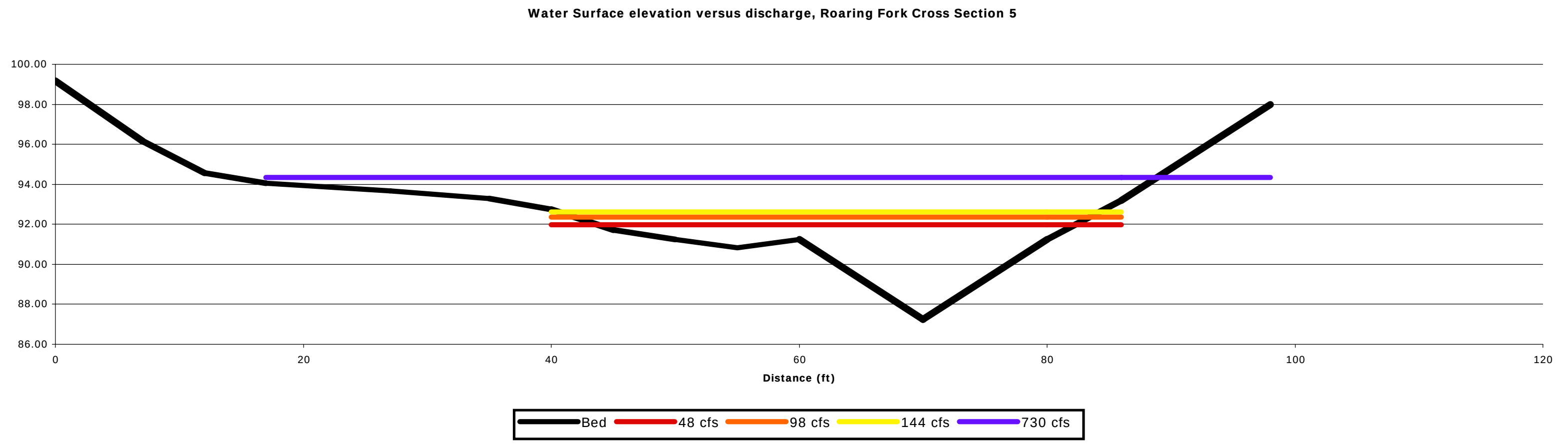

Figure 20. Comparison of water surface elevations for high risk flows and average August-September flows for RFR-TF Cross Section 5.

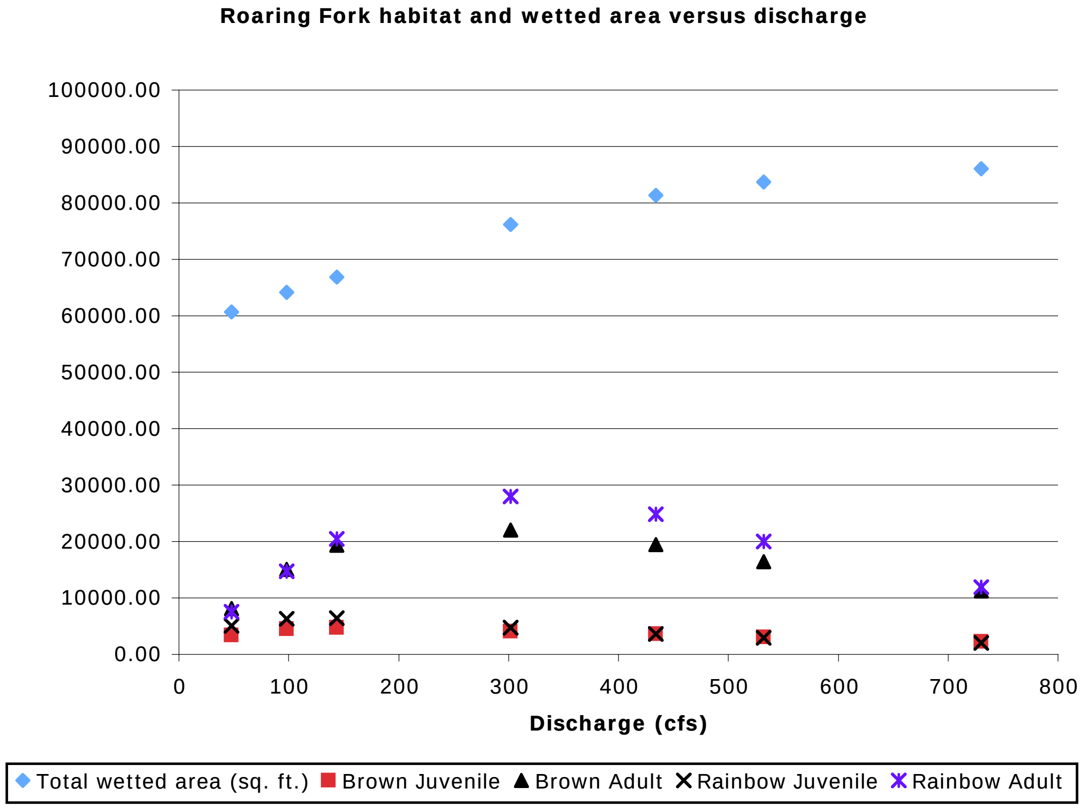

Figure 21. Total wetted area and weighted usable area for high risk and average August-September flows at Site RFR-TF.

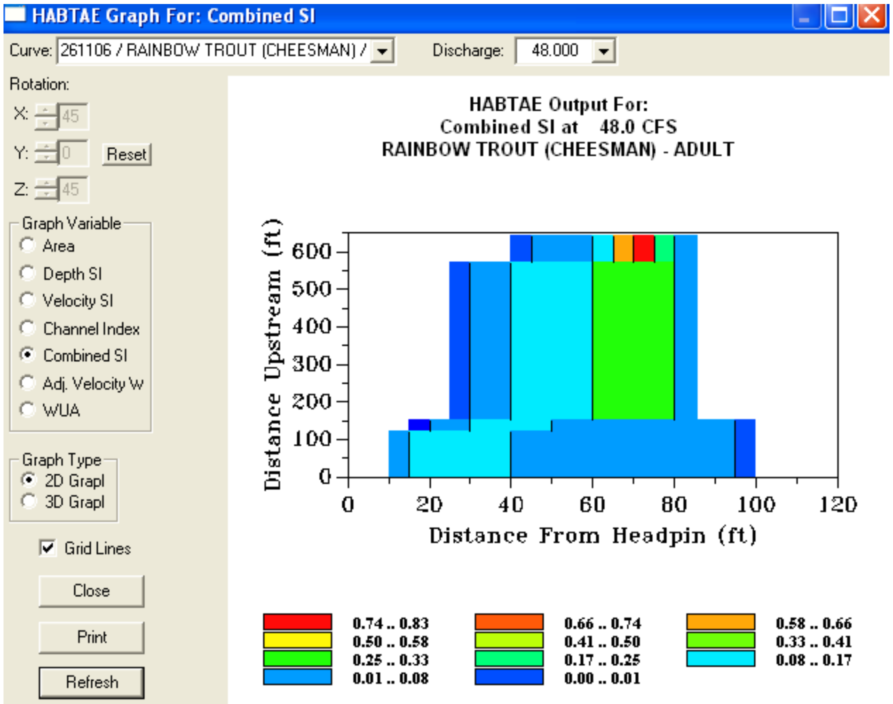

Figure 22. Example of plan view of rainbow trout habitat for Site RFR-TF at 48 cfs.

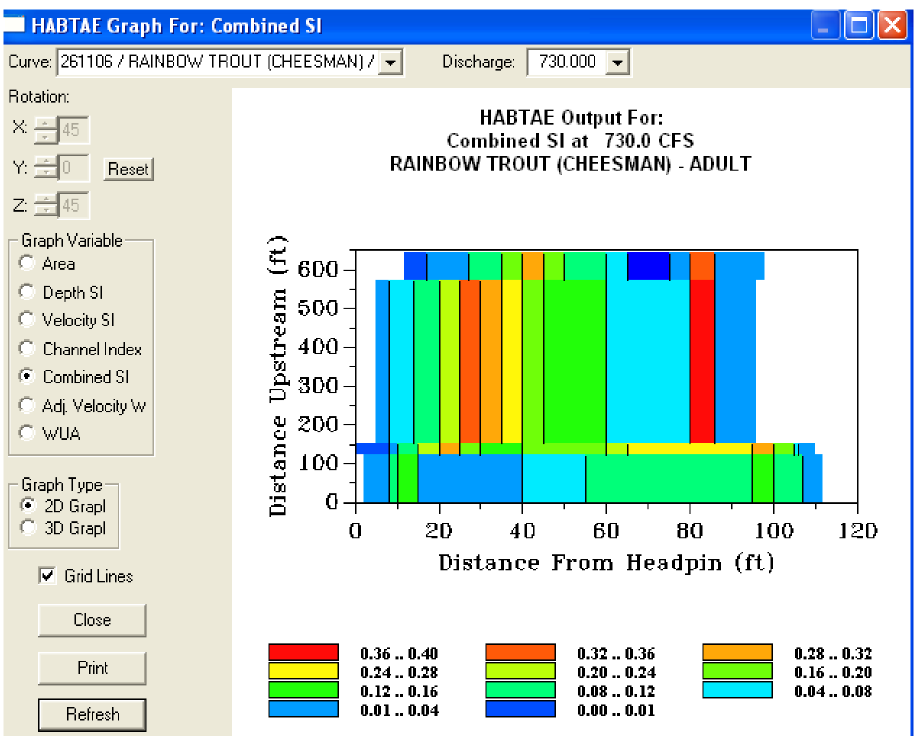

Figure 23. Example of plan view of rainbow trout habitat for Site RFR-TF at 730 cfs.

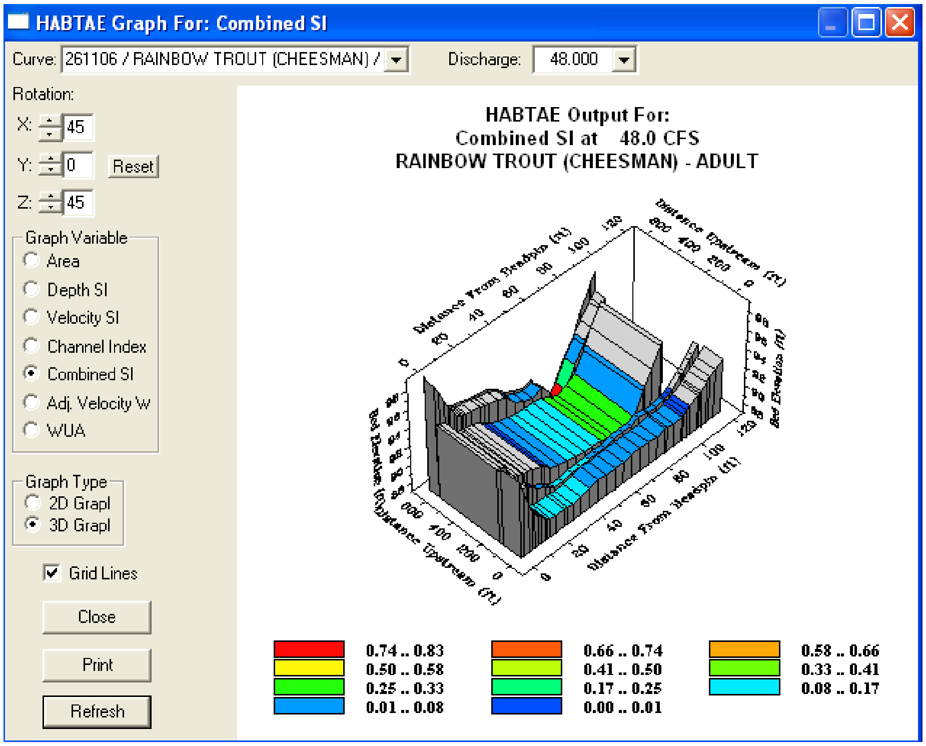

Figure 24. Example of 3 dimensional view of rainbow trout habitat for Site RFR-TF at 48 cfs.

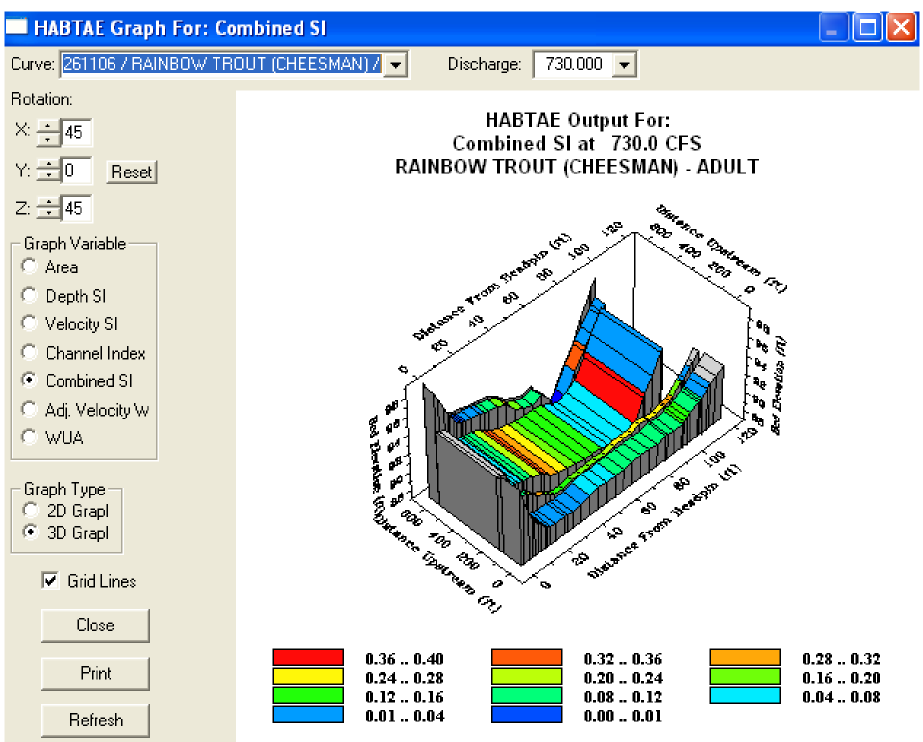

Figure 25. Example of 3 dimensional view of rainbow trout habitat for Site RFR-TF at 730 cfs.

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Appendix B

Flow-Ecology Relationships for the Watershed Flow Evaluation Tool

Colorado State University

Flow-ecology relationships for the watershed flow evaluation tool

A project funded by the Colorado Water Conservation Board

Thomas K. Wilding and N. LeRoy Poff
9/4/2008

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1. Introduction

Background

The Colorado Water Conservation Board is assisting the Interbasin Compact Roundtables with their NCNA (Non-Consumptive Needs Assessments). The NCNA will (1) identify priority areas and reaches for environmental and recreational attributes, and (2) based on Roundtable direction and needs, identify the quantities of seasonal flows necessary to maintain priority areas and reaches. A component of goal 2 is the Watershed Flow Evaluation Tool (WFET), a coarse screening tool that can be applied by stakeholders in selected watersheds to assess the potential status of biological resources under existing hydrologic conditions. WFET pilot studies are underway for the Roaring Fork River and Fountain Creek (Colorado Springs) watersheds. After the pilot study is complete, results will be shared with the Basin Roundtables who may then decide to apply the tool in their basin. The goal of this report is to develop relationships (quantitative where possible) between measures of environmental condition and levels of stream flow for Colorado. These relationships will support the development of the WFET.

ELOHA

The WFET is a specific application under the broader framework known as The Ecological Limits of Hydrologic Alteration (ELOHA). The hallmark of this new framework is that it offers a flexible, scientifically defensible approach for broadly assessing environmental flow needs when in-depth studies cannot be performed for all streams or rivers in a region (Arthington et al. 2006; Poff et al. *In Press*). ELOHA builds upon the wealth of knowledge gained from decades of river-specific studies, and applies that knowledge to geographic areas as large as a state, province, nation, or large river basin (see the TNC Factsheet for more information, TNC 2008).

Determining Non-Consumptive Flow Needs

This report is intended to assist the assessment of the potential status of aquatic, riparian and recreational resources for Colorado streams, and we are concerned with the flow-dependence of ecosystems. Flow is sometimes called a master variable because it limits the distribution and abundance of riverine species and influences other important environmental features such as water quality (Figure 1). It is important to understand the wide range of direct and indirect effects of flow on river ecosystems. The area of stream that is wet limits how many fish can survive day-to-day. This is an example of the direct importance of flow, but indirect effects may be less intuitive. Floods mobilize sand and cobbles and shape the stream-bed, and this process indirectly determines the area of pool and riffle habitat. Fast-water animals specifically require riffle habitat and fish need pools that are deep enough to avoid freezing solid in winter. Animals show an immediate response to drying or freezing, but it may take years for stream ecosystems to respond to loss of channel maintenance flows. For this reason it is easy to overlook the importance of flow beyond basic life support. Flow does not act alone in determining the types of animals and plants living in a stream. For example, steep, mountainous streams are cooler than plains rivers, and will support different communities because of these differences alone. The effects of flow change in Colorado rivers therefore need to be considered within the context of other key environmental variables, such as water temperature and channel geomorphology (shape).

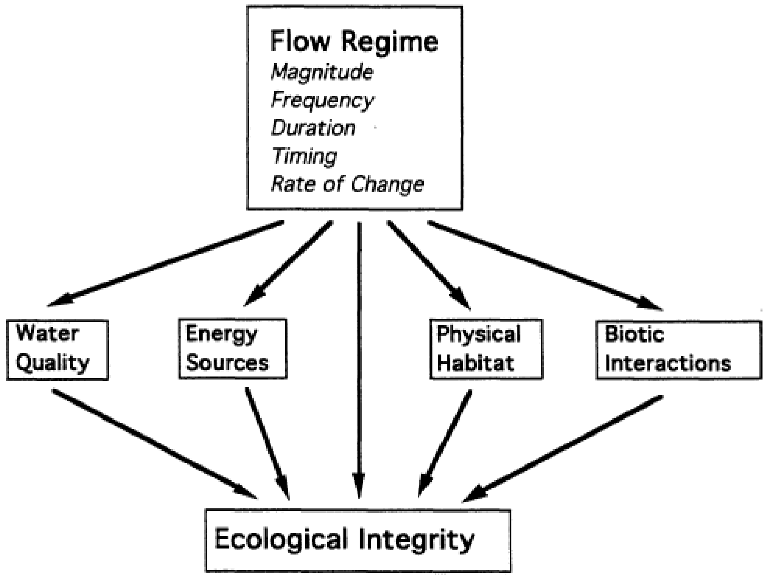


Figure 1 The flow regime, and its components, are of central importance in sustaining the ecological integrity of rivers (Poff et al. 1997).

Producing relationships that are specific to a stream type and biological community, as recommended by Arthington et al. (2006), specifically incorporates key environmental variables and therefore improves the precision of relationships with flow. Major river types in Colorado include the Rocky Mountains, Western Interior and Great Plains (see Methods section). Stream communities can be broadly classed as riparian vegetation, fish, invertebrates and so on. Each of these stream communities is sensitive to particular flow parameters such as the size of floods or the duration of extreme low flows (Richter et al. 1996). The number of combinations of stream type, community and flow parameter would be unwieldy for Colorado. We have focussed on specific combinations that represent an important component of ecosystem function, and for which data exist to provide a basis for flow-ecology relationships (published data sources). Relationships with flow are detailed in this report for riparian vegetation, stream invertebrates, warm-water fish, trout and recreation. Most of these attributes featured among valued non-consumptive uses identified by roundtable groups, with the exception of invertebrates. They were incorporated in this assessment because fish and birds depend on invertebrates for food, directly or indirectly (Allan 2007; Binns and Eiserman 1979; Jowett 1992).

Here, we describe the response of stream communities to flow change to provide a basis for analysis of non-consumptive flow needs by roundtable groups. This report provides information to answer the question: for a given change in flow, what amount of stream community response can be expected? Deciding what is an acceptable level of change, or risk of change, is a social process that can be informed by (not necessarily resolved by) the scientific information that we seek to gather (Poff et al. *In Press*). This document will enable informed decision-making about the impacts of flow alteration on non-consumptive attributes. Applied at a broad level, it is hoped this tool will aid in the identification of stream segments or subwatersheds where aquatic and riparian resources are at risk due to high water demand and, further, distinguish which non-consumptive users of the water are most at risk (e.g. cottonwoods versus trout). One application of this tool would be to identify places where more detailed site-specific investigations are needed.

2. Methods

Information on responses to flow change was sourced from a range of scientific literature (journal articles, technical reports and theses). The database of Poff and Zimmerman (*In Press*) was used as a starting point, with additional publications sought to improve coverage of Colorado stream types. Equivalent stream types from neighboring states were incorporated to bolster relationships. Literature searches were based on keywords, cross-references and scanning the publications of leading authors in each field. Discussions with relevant experts provided additional information sources and avenues of investigation. An expert-panel was assembled to provide comment on trout of the Rocky Mountains and fish communities of the Great Plains (see Acknowledgements section for participants). We did not seek endorsement or consensus from the panel, though major revisions of the draft were made to incorporate more relevant research and a broader understanding of critical issues.

Location data were extracted for most sources (Latitude and Longitude). Site descriptions were often limited but adequate for identifying the state, river system and geomorphic setting. Streams were nominally classified as one of three broad types; Rocky Mountains, Western Interior and Great Plains. Site descriptions were helpful, as was aerial photography from Google Earth ®.

Stream Types

Relationships of stream communities were investigated for individual stream types to increase the precision of relationships. Colorado was divided into three major stream types: Rocky Mountains, Interior Western and Great Plains (after Graf 2006 and Fausch and Bestgen 1997).

Great Plains rivers flow east from the Rocky Mountains, crossing the semi-arid plains. Snowmelt is a shaping feature of the natural hydrograph for mainstem rivers that have Rocky Mountain headwaters, such as the South Platte and Arkansas Rivers (Fausch and Bestgen 1997). Spring rain and summer convective-storms produce high flows (and occasional intense flood events) for all waterways of the Great Plains, with marked inter-year variability. Graf (2006) reported more variable flow regimes for Great Plains rivers compared to Interior Western rivers. Baseflows in tributary streams that originate on the plains are dependent on groundwater, which sustain perennial flow in some tributaries (e.g. historically for the Arikaree River). Channels are typically wide sandy beds (in natural settings), sometimes forming rock canyons and arroyos (incised earth channels). Historically, riparian trees may have been rare or cyclical features of those streams that lacked stable baseflows (Fausch and Bestgen 1997). The wide and sandy braided-channels of Great Plains rivers have narrowed to single thread channels, with riparian vegetation encroachment following reduced snowmelt flows from regulation (Johnson 1994).Rocky Mountain streams have a strong snowmelt signature, clear waters and generally coarse stony substrates. Summer temperatures are relatively cool and stream gradients are steeper than both Interior Western and Great Plains streams. Headwater streams in the western half of Colorado are predominantly Rocky Mountain streams, with example rivers including the Roaring Fork, Cache Le Poudre (above Fort Collins), Big Thompson and Fountain Creek headwaters.

Interior Western streams are characterized (in natural settings) by warm temperatures in summer, high turbidity, and a geomorphological setting that varies from canyons to alluvial floodplains. A degree of snowmelt-runoff pulses through Interior Western rivers (often sourced from Rocky Mountain headwaters), with increasing contributions from arid, highly-erodible landscapes further downstream. Example rivers include the non-headwater sections of the Colorado, Gunnison and Yampa.

Defining the boundaries between the above stream types is outside the scope of this report. For terrestrial systems, it is sometimes adequate to draw lines on maps to delineate ecosystem classes. Rivers are more accurately viewed as a product of the entire watershed, and hence they make gradual transitions with inflows from different land systems and changing geomorphic settings (Snelder et al. 2005). As an example, Fausch and Bestgen (1997) describe a transition-zone for rivers flowing from the Rocky Mountains out onto the Great Plains. Along the front-range, these rivers and streams feature cool temperatures, cobble substrates and single-thread channels of moderate gradient that are shaded by riparian trees. Sections of Fountain Creek presumably fall into this category. A similar transition is expected between the Rocky Mountains and Interior Western area.

Metadata

The literature review was extensive, covering a broad field of research. Some disciplines have received more attention from researchers, and Table 1 provides a breakdown of studies by community and stream type. These numbers represent studies relevant to flow change, though not all were ideal for deriving flow-ecology relationships (see Results section). Only 34 studies were actually undertaken in Colorado. The remainder (from Wyoming, Utah, Kansas, etc.) represented equivalent stream types and were critical in achieving adequate sample sizes.

Some of the more intensively studied areas include riparian vegetation and fish of Interior Western rivers. More intensive research on the Green, Yampa and Colorado Rivers encompass much of this work. Riparian vegetation is also the focus of many studies on Great Plains streams, together with fish. Studies of Rocky Mountain streams more often focus on invertebrates and fish.

Table 1 Number of studies contributing to this report, broken down by stream type and community. The "other" category includes studies from non-Colorado stream types and other communities. This report focussed on areas highlighted in bold.

	Interior Western	Rocky Mountains	Great Plains	Total
Fish	**19**	**18**	**15**	50
Riparian vegetation	**20**	1	**8**	28
Invertebrates	9	**9**		18
Vertebrates (birds, beaver)	4			4
Terr. Invertebrates	2		1	3
Algae	2			2
Total	56	25	24	105
Other				44

Data Analysis

Within the confines of available data formats, some general guidelines were followed in determining what should represent an individual response (i.e. data points for flow-response plots). For example, sites were used as individual data points for diverse invertebrate communities, with community information summarized using abundance and diversity metrics. The response of less diverse communities was sometimes represented using individual species as data points (e.g. biomass of Colorado pikeminnow).

Ecological responses were limited to species that are indigenous to the area of study, so excluding the response of potential pest species (e.g., tamarix). Combining the two groups in the same plot would complicate interpretation of the output in terms of assessing responses of valued native species or community types (e.g., cottonwood). Trout were an exception to this rule. Introduced species (brown, rainbow and brook trout) were included in the response analysis because of their recreational value.

The flow parameters used by researchers were not consistent across the literature. Our investigations focused on the effects of peak flow and low flows. Duration, magnitude and timing were occasionally reported in the source literature but not often enough to derive relationships. We attempted to standardize peak flow to 24-hour average annual peak flow, helped by the consistent use of this parameter in many studies. Likewise, estimates of low flow were typically standardized to 24-hour average annual low flow (this was sometimes limited to the summer/autumn period). Dividing flow by watershed area or mean annual flow to produce a specific discharge was attempted where percent flow alteration (relative to a pre-management baseline) was not used. Producing relationships both derived from and applied to different sized rivers can benefit from standardization by some correlate of channel size (mean annual flow or, failing that, watershed area). Although attempted for low flows, only relationships with peak flow benefited from such standardization.

By comparing measured ecosystem parameters across a range of flow conditions (varying levels of modification), emerging patterns provide a basis for quantifying ecosystem response (Figure 2). Analysis methods were tailored to suit the available data, and these are described in the results for each community. The mechanisms by which flow alteration affect stream ecosystems are complex, so a simple response to flow (1-dimensional) was not anticipated. A community could be limited by the chosen flow-parameter (e.g. peak-flow), but other parameters (sometimes unmeasured) often constrain the ecosystem and limit its response to flow. For example, cutthroat trout may reach higher biomass in deeper channels, but if introduced competitors (brook trout) are present then the trout population will be small regardless of depth (Dunham et al. 2002). Using quantile regression to define the upper bound is therefore expected to better represent the potential response to the chosen flow parameter (see Cade and Noon 2003). This also expresses complex relationships in an easily digestible form for end-user application, as compared to multi-dimensional models.

Quantile regression was used to identify these upper bounds, providing a coarse filter to isolate the potential response to each flow parameter (using Blossom statistical software, Cade and Richards 2007). This method minimizes the sum of absolute deviations (least absolute deviation), which are asymmetrically weighted by the quantile (e.g. 90%) for positive residuals and one minus the quantile for negative residuals (e.g. 1-0.9=0.1). Using absolute deviations (cf. squared deviations for conventional regression) reduces the effect of outliers. In most cases, 90% quantiles were judged as representing the upper-bound response adequately. Transformations were applied to the data, as necessary, before carrying out linear quantile regression.

The significance of the relationships was tested (null hypothesis: slope =0) using a quantile rank score test to minimize assumptions regarding error distributions (cf. higher power parametric alternatives). The rank score test provides P-values that are calculated from the sign of the residuals (+ve or –ve), not their magnitude. The permutation version uses an F statistic with its sampling distribution approximated by permutation (Cade et al. 2006), with 1000 permutations used here. In cases where both flow and the response parameter were quantified as a percent-change, relative to some reference condition, the equation intercept was assumed to be zero (where zero must be the reference condition for each data point).

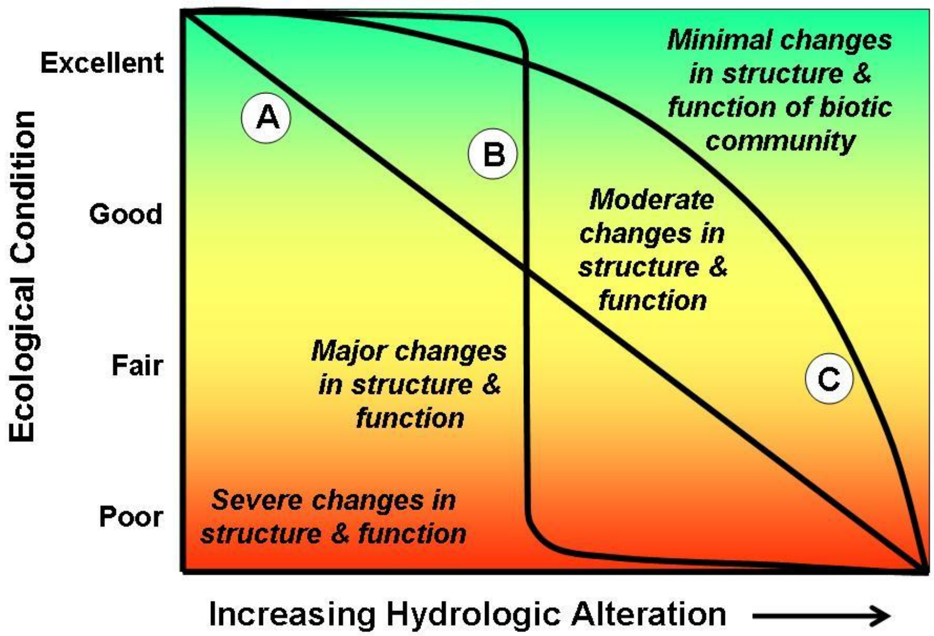


Figure 2 Three of several possible forms of flow alteration-ecological response relationships: linear (A), threshold (B), and curvilinear (C). The form of the curve depends on the specific ecological and hydrological variables analyzed. (Adapted from Davies and Jackson 2006).

3. Results

Riparian Vegetation

The response of riparian vegetation to changes in peak flow could provide a basis for generalized flow needs (Figure 3). Interior Western rivers (black dots, Figure 3) are consistent with the bounded response, but depend on riparian data from other river types (Great Plains or non-Colorado) to define the limits of the response over a wider spectrum of alteration. Different types of vegetation display a varied response to reduced disturbance by peak flows. For example, wetland plants can increase under stable flow conditions without periodic floods (Merritt and Cooper 2000), providing a numerically positive response to reduced peak flows. Conversely, cottonwood numbers respond negatively to a lack of flood recruitment events (Lytle and Merritt 2004; Richter and Richter 2000). Combined, the positive and negative numerical responses to flow alteration signify a shift in community composition from the natural riparian forest, and these can be plotted as an absolute (positive) percent change (Figure 3). The absolute response is therefore a coarse representation of the complex effects of flow change. Not all species and populations are expected to be equally sensitive to flow change, so the 90% bound (quantile regression line) provides a delineation of those species that are vulnerable. The 90% quantile line approaches a 1:1 relationship for the response of riparian vegetation to peak flow alteration. This describes, for example, that a 50% change in peak flow could produce up to a 60% change in riparian vegetation (with 10% probability of greater effects). No Rocky Mountain rivers were included in the dataset, but they are expected to show a similar response. For example, willow establishment responded positively to peak flow magnitude (> 2-year return period flow) for Rocky Mountain streams with natural flow regimes (Cooper et al. 2006).

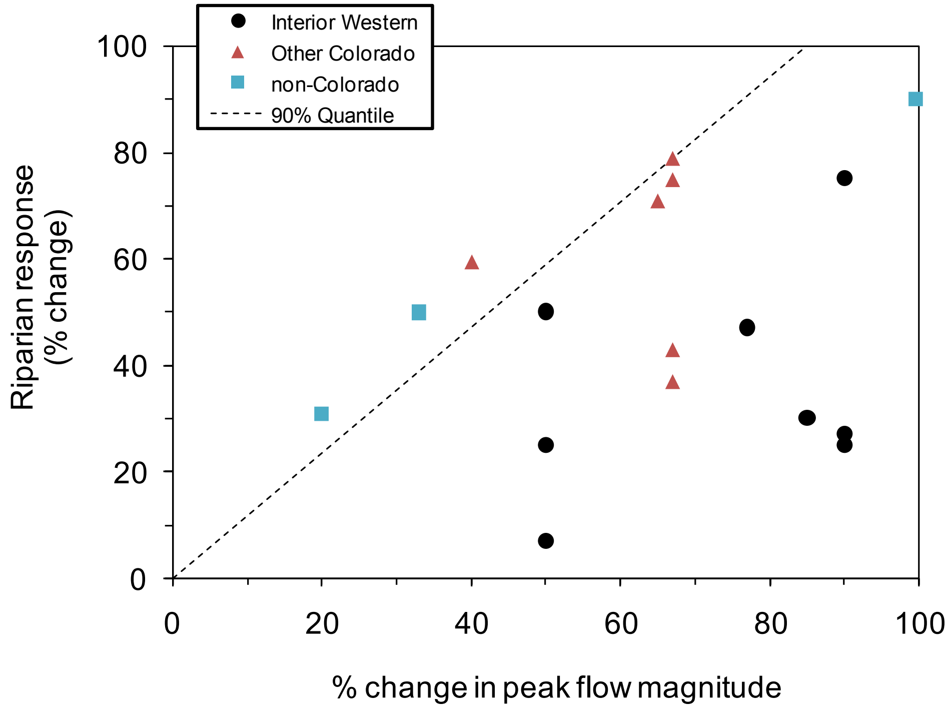
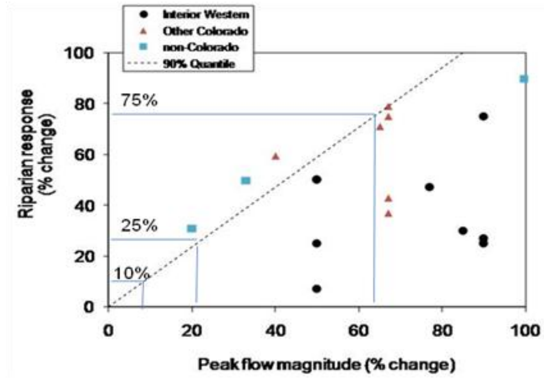


Figure 3 The response of riparian vegetation to change in peak flow. Riparian response is the percent change in riparian metrics relative to a reference condition. Percent change in peak flow is also relative to a reference condition (typically a reduction in snowmelt peak). The response for Interior Western rivers is reinforced by a similar response from riparian communities elsewhere. Quantile regression provides a 90% bound on the response (Y = 1.18 * X; forced zero intercept to reflect no riparian response to zero change in flow).

The relationship portrayed in Figure 3 indicates that the greater the change in peak flow, the greater the risk of a change in riparian vegetation (deviation from reference condition). To minimize the risk of a change in riparian vegetation, end-users might decide a small change in riparian vegetation is acceptable (e.g. 10%), and then use Table 2 to determine the corresponding change in peak-flow (8% in this case). Likewise if the acceptable level of riparian change is 50%, then the corresponding flow change is 42% (Table 2). This allows the end user to decide the level of risk that is acceptable. The riparian response values in Table 2 are based on the upper-bound response (90% quantile) to represent those populations that are susceptible to a change in peak flow. This minimizes the number of populations that will show a greater response (100% - 90% = 10% of populations, in this case).

Table 2 The change in peak-flow expected to produce five levels of riparian response (% change). This is calculated, based on data presented in Figure 3, for a 90% quantile (upper bound). The plot to the right shows the derivation of three points from the quantile regression, as an example.

Riparian Response	Peak-flow change
10%	8%
25%	21%
50%	42%
75%	64%
90%	76%



In addition to peak flow magnitude, the timing of peak flows and rates of recession are also important for maintaining riparian forests, as this determines seedling mortality (Cooper et al. 1999). Specifying hydrographs to this level of detail is beyond the scope of this document and should instead be incorporated into site-specific studies where riparian vegetation is a critical issue.

Riparian vegetation responds to peak flows (via sediment supply, disturbance, seedling establishment, etc.), but can also respond to low flow. Seedling reliance on surface water continues after seasonal peak flows have receded (Cooper et al. 1999), but extended periods of zero flow are required (>24% of the time) before changes in riparian vegetation are seen (Lite and Stromberg 2005). As flows approach zero, groundwater levels will determine water availability for riparian vegetation. This is perhaps why the majority of studies focus on response to groundwater levels, rather than river flow (Cooper et al. 2003; Scott et al. 1999; Stromberg 2001).

Significant changes in riparian vegetation are often observed where annual low-flows have actually increased due to dam operations (Merritt and Cooper 2000; Shafroth et al. 2002). The elevated low flows may increase survival of some species; however, coincident decreases in peak flow and sediment supply make it difficult to quantify what appears to be a secondary response to low flow. The importance of groundwater levels and peak flows for sustaining riparian vegetation is well established. Riparian response to change in low flow may not be a critical issue, compared to fish and invertebrate response, and so is not quantified here.

Stream Invertebrates – Rocky Mountains

Most of the invertebrate data are from Rocky Mountain streams with flow diversion structures. Here we have the luxury of a large number of diversion sites, evaluated using standard methods, with few confounding effects. By drawing data from two studies meeting these criteria (Albano 2006; McCarthy 2008), more subtle responses can be distinguished. There is a range of metrics available for summarizing invertebrate data, many of which represent the pollution tolerance of sensitive species in the community. However, when evaluating community response to flow change, the diversity and abundance of invertebrates that require riffle or fast water habitat (so-called obligate rheophiles, as designated by Poff et al. 2006) are more appropriate indicators. For Rocky Mountain streams, this group of invertebrates responded to large reductions in flow (Figure 4). This is more apparent from the McCarthy (2008) data, which focussed on a uniform group of small subalpine streams in the Fraser River basin (elevation 7,500 to 13,000 feet).

Diversity of rheophiles may show a threshold response to flow quantity (i.e. response to a specific flow rate, rather than a percent change in flow), with declining diversity below about 0.2 cfs (Figure 5). The potential density of rheophile species increased with flow (Figure 6), particularly for flows less than 2 cfs. In terms of the amount of food available to predators, such as fish and birds, density of invertebrates (number per unit area) is important, but the total number of invertebrates can limit the number of predators supported by a stream. This means a larger area of stream with the same density of invertebrates can potentially support more trout (total number of invertebrates = density x area). The area of wetted stream increases with flow beyond the thresholds mentioned above, and site specific studies could be used to describe this relationship (e.g. wetted perimeter or PHABSIM). Alternatively, the low-flow categories from Binns and Eiserman (1979) were partly based on habitat area measures and hence may represent the response of total invertebrate production to flow, if greater than 2 cfs (see Table 3 for categories).

Alteration of peak flows can affect invertebrates because flood disturbance is important in limiting algal growth (a major food source for scrapers) and maintaining habitat. Peak flows also represent a direct disturbance to invertebrates. Consequently, high disturbance streams have contrasting biota to low disturbance streams (Lytle and Poff 2004). The dataset used to evaluate the effects of baseflow (from McCarthy 2008 and Albano 2006) was not suitable for reviewing the effects of peak-flow alteration because flow was measured at the time of sampling only and not during peak flow.

Resorting to the larger Colorado dataset necessitates a broader view of the invertebrate community than just the obligate rheophiles, and consequently a more variable response is seen (Figure 7). The large response of some invertebrate metrics to reduced flood disturbance produced a strongly skewed dataset. This may be more pronounced for Interior Western rivers (e.g. density increased by an order of magnitude below Flaming Gorge Dam, Vinson 2001), where natural extremes in temperature and turbidity are potentially cut by flow regulation. Because of the skewed response, a 75% quantile was considered a more comparable indicator of response across Rocky Mountain and Interior Western streams, compared to the 90% quantile used for other datasets (Figure 7).

Timing of peak flows and water temperature are important seasonal triggers for the life cycles of many invertebrates (Lytle and Poff 2004), but are not dealt with in this report.

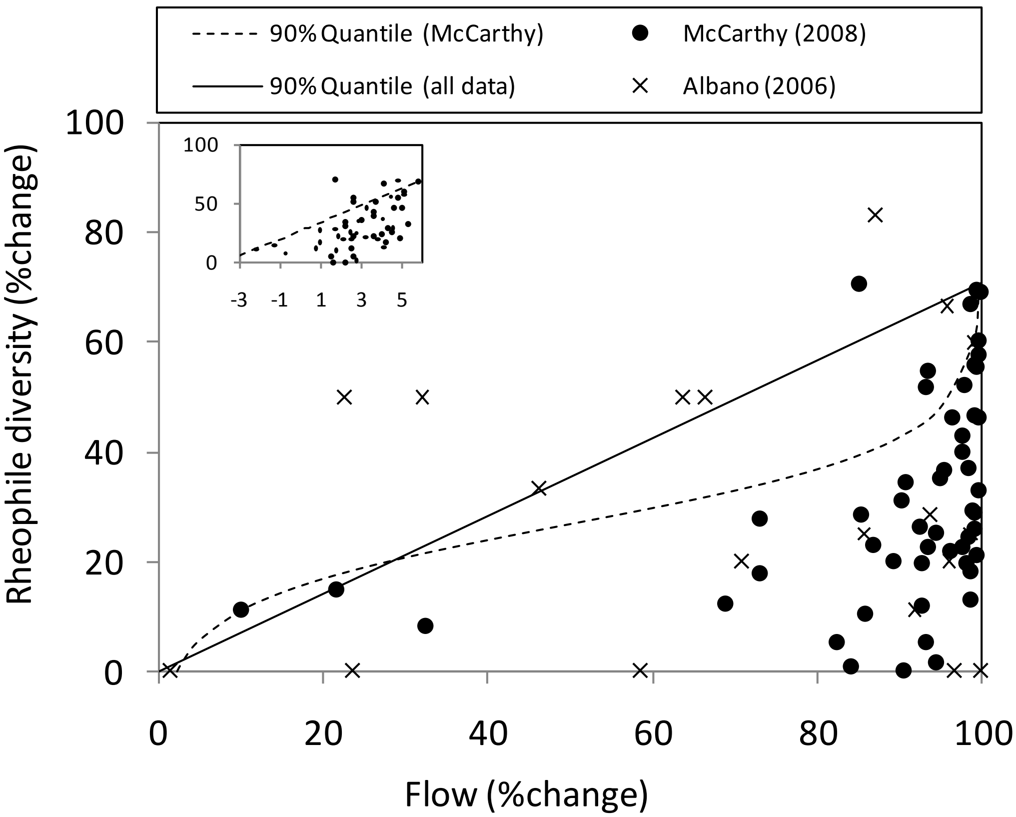


Figure 4 Response of Rocky Mountain invertebrates to change in flow (flow reduction downstream of the diversion on the day of sampling). Invertebrate response is measured as % change of rheophiles (fast-water species), compared to reference sites upstream of the diversion. Data from two literature sources are presented (Albano 2006; McCarthy 2008). Upper bounds for the data are represented as 90% quantiles. The quantile for the McCarthy (2008) data was calculated after transformation (using logit for %flow reduction) to better represent the skewed response (response function; $Y = 7.2 * \mathrm{Ln}(X/(100-X)) + 26.77$, P-value = 0.0839). The inset shows the McCarthy data plotted on a logit scale. Three data points are not shown to clarify the core data pattern (Albano study x,y; 63,300; 91, 400; McCarthy 111,14).

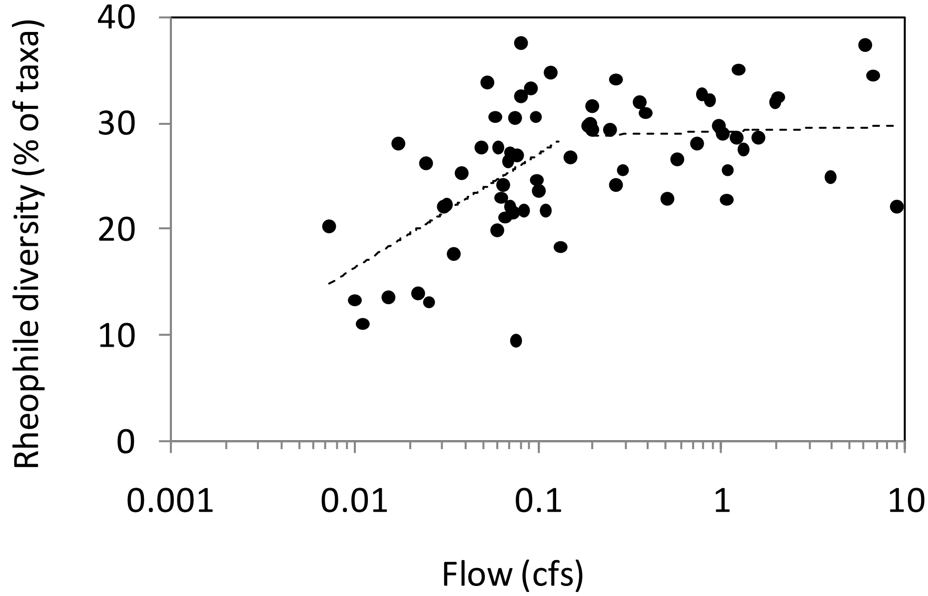


Figure 5 Response of rheophile (fast water) invertebrates to flow (log scale). The number of rheophile species is expressed as a percentage of total number of taxa per sample. Regression lines (log) are fitted to two separate bins of data to illustrate an apparent threshold response. Data are from McCarthy 2008).

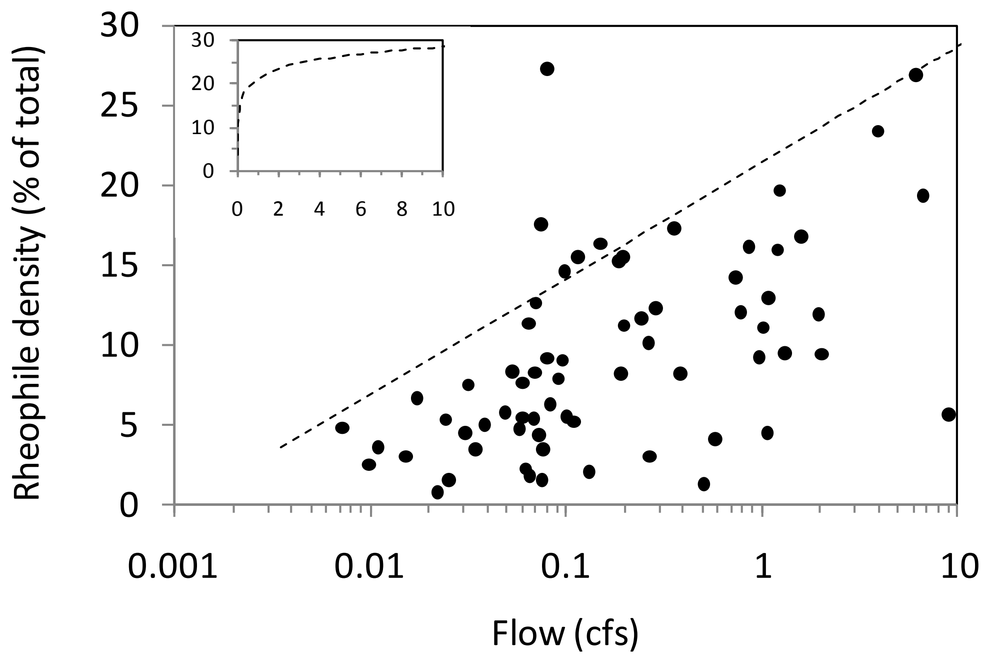


Figure 6 Response of rheophile (fast water) invertebrates to flow (log scale). The density of rheophile taxa are expressed as a percentage of total sample density. Data are from McCarthy 2008). The upper bound for the data is represented as a 90% quantile ($Y = 7.24 * Log_{10}X + 21.4$; $p = 0.001$), and the inset plot shows this on a normal (arithmetic) scale.

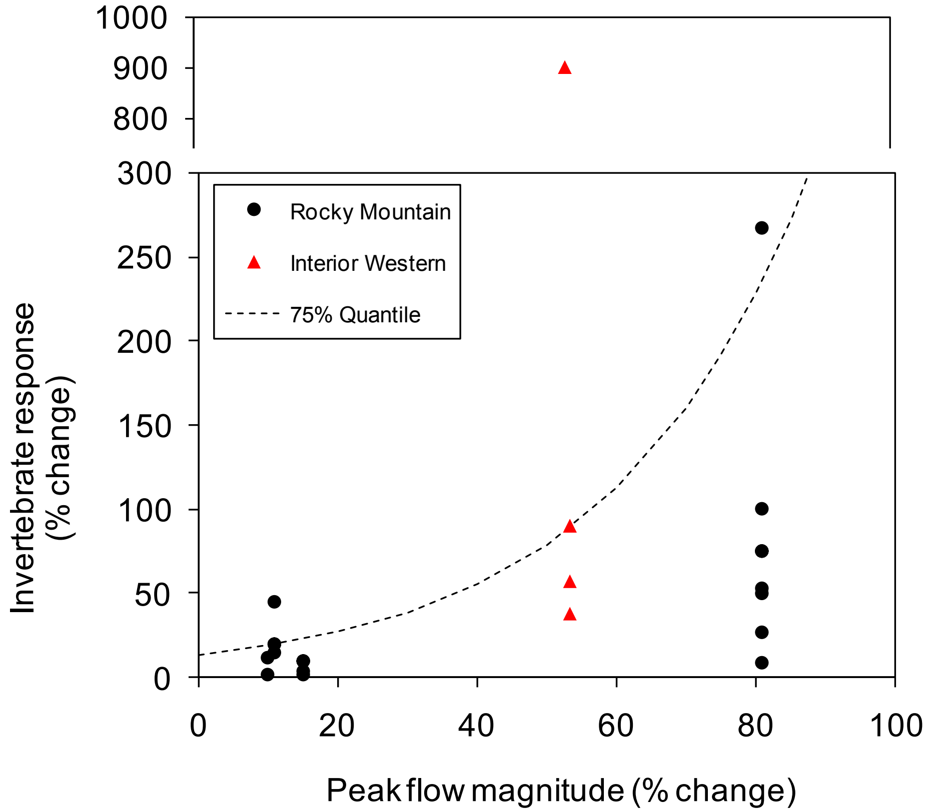


Figure 7 Response of stream invertebrates to a reduction in peak flow. The upper bound for all data is represented as a 75% quantile ($Log_{10}Y = 0.015 * X + 1.13$; $P = 0.123$). Note the discontinuous y-axis (higher scale > 300%).

Cold-water Salmonids – Rocky Mountain area

Several salmonid species are found in cold water streams and rivers of Colorado, including native cutthroat trout and three introduced species (brown trout, rainbow trout and brook trout). The introduced species represent an important recreational fishery in Colorado. Trout distributions can be explained in terms of water temperature (both upper and lower altitude limits) and interactions among species (competitive, predation). Requirements for cool temperatures create a lower altitudinal limit that largely confine trout fisheries to the Rocky Mountain area. But dams that release cool water to otherwise warm water rivers (Interior Western), sustain excellent fisheries (Merwin 2008).

An inability to reproduce successfully during short summers is expected to set the upper altitudinal limit for trout (Coleman and Fausch 2007). The order of cold-tolerance (stenothermy, from cold to warm) is cutthroat, brook, rainbow and brown trout (Raleigh et al. 1986). The order of competitive advantage is the reverse, which often excludes cutthroat and brook trout from lower elevation waters where temperatures are otherwise tolerable (McHugh and Budy 2005).

Within the confines of their temperature range and competitive exclusions, the abundance of salmonids is potentially limited by flood disturbance during critical life stages. In Colorado streams, this bottleneck is the magnitude of snowmelt coinciding with fry emergence (Fausch et al. 2001; Latterell et al. 1998; Nehring and Anderson 1993). The timing of fry emergence varies from year to year, between species and with altitude. But generally speaking, brown trout fry emerge during May and June for Colorado, with rainbow trout fry emerging in late June and July (Nehring and Anderson 1993). With snowmelt runoff often peaking in June for Rocky Mountain streams, the potential for snowmelt to overlap with fry emergence is high for brown and rainbow trout. Brook trout emerge earlier (fall spawners), but they are still vulnerable to flow disturbance (Latterell et al. 1998). Cutthroat trout fry emerge later in summer, and this native species appears better adapted for avoiding disturbance from snowmelt runoff.

Several authors have documented the negative correlation between peak flow magnitude and recruitment success in trout (Fausch et al. 2001; Latterell et al. 1998; Nehring and Anderson 1993). Data was sourced from technical reports by Nehring and Anderson (Nehring 1986; Nehring and Anderson 1985) to generate quantitative relationships for brown and rainbow trout (Figure 8). To describe relationships across multiple sites, individual site-year values for density of juvenile trout (number of age 1+ trout per unit area) were standardized by the maximum value for that site, and peak flow (monthly average) was standardized by the mean annual flow. Rivers monitored include the Arkansas, Gunnison, Rio Grande, South Platte and Cache la Poudre (mean annual flow ranged from 170 to 1400 cfs).

Density of juvenile brown trout declined steeply with peak-flow (Figure 8). The lower bound (10% quantile) gave a better response to peak flow than the upper bound (P-value 0.016 and 0.094 respectively). This suggests flow disturbance has a more consistent effect on trout recruitment in otherwise bad years (i.e. when unmeasured parameters are unfavourable). In an otherwise good year for recruitment, peak monthly flows of up to 4 times mean annual flow can still produce high recruitment (from the upper bound, Figure 8). In an otherwise bad year, 2 times mean annual flow will be sufficient to limit recruitment to less than a third of maximum (from the lower bound response). There is a greater risk of recruitment failure if the average flow for a month exceeds 6 times mean annual flow (Figure 8).

June was typically the month with the highest average flow (peak snow melt), but July flows produced a better correlation with juvenile brown trout (R^2 values for April to September respectively: 0.107, 0.220, 0.342, 0.547, 0.307, 0.451). It is not clear whether this reflects a higher susceptibility of juvenile trout to disturbance in July, or perhaps July flow better captures the duration of disturbance acting on that year class (i.e. high snow melt extending well into July). Similar conclusions regarding recruitment limitation would probably be drawn, whether predictions are based on peak flow (monthly average) or July flow, because of the similar form of the relationship (compare Figure 8 and 9).

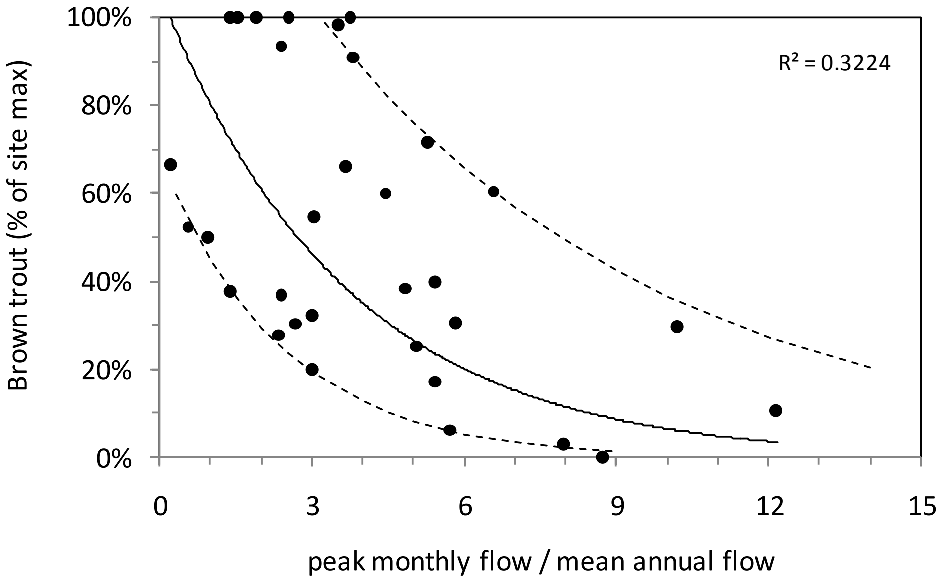


Figure 8 Response of juvenile trout to peak-flow. Recruitment success is measured in terms of the density of age class 1+ brown trout, and is standardized by the observed site maximum. Peak monthly flow was standardized by mean annual flow. The data are sourced from Nehring 1986 and Nehring and Anderson 1985. A standard regression line (solid line, exponential) and corresponding R^2 value is presented, along with 10% and 90% quantile regression lines (dashed lines, P-value 0.016 and 0.094 respectively), fitted to Log_e transformed trout data.

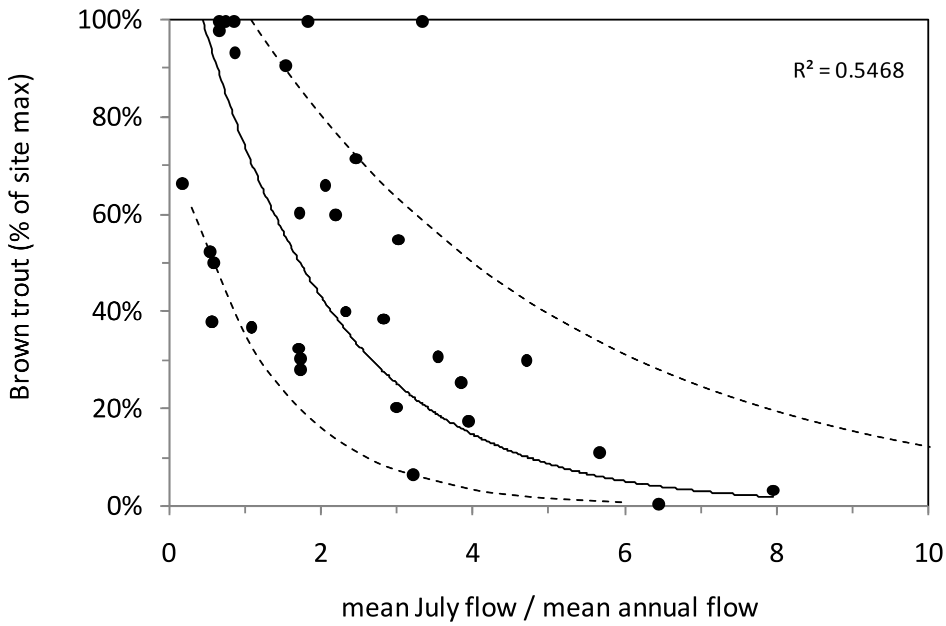


Figure 9 Response of juvenile trout to July flow. Recruitment success is measured in terms of the density of age class 1+ brown trout, and is standardized by the observed site maximum. The average flow for July was standardized by mean annual flow. The data are sourced from Nehring 1986 and Nehring and Anderson 1985. A standard regression line (solid line, exponential) and corresponding R^2 value is presented, along with 10% and 90% quantile regression lines (dashed lines, P-value 0.002 and 0.089 respectively), fitted to Log_e transformed trout data.

Rainbow trout data were also collected for the same studies (Nehring 1986; Nehring and Anderson 1985), but this species was recorded at lower densities and at fewer sites. The data were adequate to describe a similar decline in the density of juvenile rainbow trout with increasing flow, which was most pronounced for July (exponential decay, $R^2 = 0.391$). A similar response to peak flow is expected for brook trout, with (Latterell et al. 1998) describing a decline in the recruitment of trout in streams dominated by brook trout (relationship reproduced in Figure 10). It is possible that native cutthroat trout are less sensitive to the magnitude of snowmelt, given their later spawning.

The relationships describing juvenile trout response to peak flow are useful in assessing the potential effect of flow change on trout recruitment, but should not be used to imply reduced peak flows are always better for sustaining trout. High value trout fisheries can be degraded by excess recruitment of juveniles, because increased competition can reduce the average size of adult fish (Bohlin et al. 2002; Keeley 2001). The density of adult trout must be considered at this point. Rivers with abundant adult trout are more likely to experience negative effects from competition with increased recruitment. Conversely, increased recruitment is more likely to be beneficial in rivers with low densities of adult trout. Peak flows are essential for channel maintenance, including flushing of spawning gravels and food-producing riffle areas (Poff et al. 1997). Several members of the expert panel suggested the benefits of a sustained loss of peak flows may only last a few years, until habitat degradation and competition produces a net impact on the fishery. Maintaining inter-annual flow variability is therefore viewed as important for productive trout fisheries.

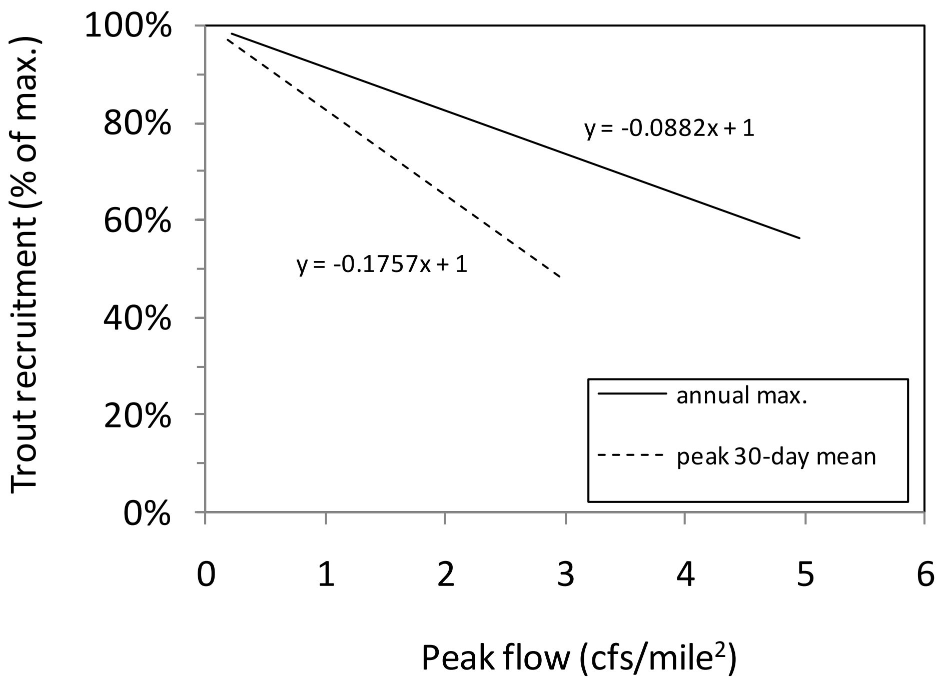


Figure 10 Response of trout recruitment to peak-flow. Recruitment success was measured in terms of the density of 1-year old fish (both brook and brown trout), and is standardized here as a proportion of theoretical maxima. Two measures of peak flow are used: annual peak-flow (24-hour average) and mean of the highest 30 days of flow (to incorporate duration). Flow (cfs) was divided by watershed area (square miles) to provide a comparable measure of disturbance for different sized rivers. Equations are derived from Latterell et al. (1998) after unit conversion for a scenario of 100 adult trout per 250 m (adult trout are a second factor in the model, but do not change the gradient of the response).

Minimum flow requirements for trout are well documented using site specific methods, such as IFIM (Raleigh et al. 1986) and empirical models (for a review, see Fausch et al. 1988). Studies of trout in Rocky Mountain streams generally identify low flow as a potentially limiting factor where temperature is otherwise suitable (Binns and Eiserman 1979; Jowett 1992; Nehring and Anderson 1993; Rahel and Nibbelink 1999; Raleigh et al. 1986). Low-flow relationships for trout can be assessed within this context by limiting application of guidelines to streams and rivers that are known to sustain trout or recreational trout fisheries. Two separate issues arise for low flows: habitat during the summer autumn period and ice refuge during winter.

Flows during winter deserve consideration as trout overwinter successfully only in pools that do not freeze to the bottom and where gill abrasion from frazil ice can be avoided (see page 106 in Annear et al. 2004). Hubert et al. (1997) outlined some of the difficulties in setting and applying flow standards that maintain refuges from ice. More recent advances in hydraulic modelling have enabled predictions of the change in habitat with flow under the ice (Waddle 2007) but are still unable to predict the effect of flow on ice formation. Complications arise at multiple scales. For example, pools can develop an even cover of ice compared to fast flowing areas that freeze along the edges to form an open tube (Waddle 2007). At larger scales, a reverse altitude effect can occur, with snow pack providing insulation and reducing ice formation in higher altitude streams (Hubert et al. 1997). In addition to low flows during winter, peak flows throughout the year are important in the formation of deep pools, and these pools subsequently provide over winter refuge areas. Access for fish upstream or downstream to ice refuges (e.g. large pools or beaver dams) will also be important. Despite the potential importance of this issue, quantitative relationships between flow and over winter survival of trout cannot be produced at this time.

The second issue arising from low flows is the amount of habitat available during summer and autumn. An earlier model of trout abundance in Wyoming streams rated the relative suitability of summer low flows as part of a broader Habitat Suitability Index (Binns and Eiserman 1979). The study included Rocky Mountain streams (most >6000 feet altitude, mean annual flow 25 to 500 cfs) and summed the abundance of four trout species (same as those found in Colorado). The authors assigned five categories for suitability of summer low flow (Table 3). The categories appear to be subjective, but they did form the basis of what remains one of the more robust predictive models for trout biomass (Raleigh et al. 1986).

The origins of the Binns and Eiserman (1979) category thresholds include earlier work by both Wesche and Tennant. Publications by Thomas Wesche dating back to 1973 document 25% of mean annual flow as a threshold of physical habitat deterioration in small trout streams in Wyoming (mean annual flow 30 cfs) (http://library.wrds.uwyo.edu/wrs/wrs-37/abstract.html). Hubert et al. (1997) cites Burton and Wesche (1974) as finding six streams where 25% of MAF was met or exceeded 50% of the time (July to September) had good trout fisheries (246-705 fish/acre). This was compared to five streams that did not meet the criterion, where trout populations were small (5 to 190 fish/acre).

Similar categories were developed by Tennant (1976) for Montana, Wyoming and Nebraska streams (see Table 5 in the section on Warm-water fish - Great Plains). This method was reviewed by Mann (2006) for its representation of physical habitat (depth, velocity, width and weighted usable area). Mann concluded that Tennant's categories provide a reasonable representation of Interior Western streams (termed the Temperate Desert Division) and of low gradient streams (<1%) such as the Great Plains area (from Nebraska correlations). In other areas, such as the Rocky Mountains (termed Temperate Steppe Regime Mountains), Mann (2006) recommended that application of the Tennant method be limited to flow standards for initial planning (i.e. should not be used to prescribe flow requirements). By comparison, the categories from Binns and Eiserman (1979) were developed for steep gradient rivers (mean slope 2.2%, range 0.1 to 10%, median 0.95%) and are considered more applicable to Rocky Mountain streams.

The categories are valid for assessing suitability of low flows if applied to temporal comparisons (i.e. changes over time for individual watersheds), or spatial comparisons across one stream type (Rocky Mountain snowmelt hydrograph streams). But spatial comparisons across different stream types may be invalid because a higher proportion of mean annual flow could simply represent a naturally stable flow regime. Its validity also depends on the use of natural mean annual flow as the reference condition for both pre- and post-development (see page 160 in Annear et al. 2004).

One drawback of the flow categories (both Tennant and the Binns and Eiserman approach) is that these may underestimate flows for small streams. The assumption that habitat for both small streams and large rivers can be represented by the same proportion of mean annual flow may not hold true (Jowett 1997). Hatfield and Bruce (2000) predicted the flow providing maximum habitat for large adult trout (modelled using PHABSIM) from mean annual flow (Figure 11). Their results demonstrate the higher flow requirements (proportionately) to maximise habitat in smaller streams. The categories of Binns and Eiserman (1979) were developed from surveys of streams as small as 30 cfs (mean annual flow), so streams too small to be represented by their categories are likely to be too small to support a recreational fishery. Smaller trout can persist in smaller streams, but flow magnitude will ultimately limit the trout biomass that sustains recreational fisheries (Jowett 1992). It is therefore important to limit the application of the categories in Table 3 to existing trout fisheries, and to calculate low flows as a percentage of natural mean annual flows (cf. altered flows).

More recent methods for assessing low flow guidelines of trout at a regional scale would require further work for application to Colorado, hence are beyond the scope of this report. Generalized habitat models were developed for New Zealand and France (Lamouroux and Jowett 2005) and offer a worthwhile avenue of research for Colorado. This method does not produce flow guidelines, but the relationships between habitat and flow that are produced may provide a useful basis for refining guidelines for low flows. Alternatively, generalized flow guidelines could be developed based on existing habitat survey results for individual stream types in Colorado, by adapting the method used by Wilding (2007).

Table 3 Categorical rating of low-flow suitability for trout (cutthroat, brook, brown and rainbow), from (Binns and Eiserman 1979). Summer flows (average for August to mid-September) are expressed as percentage of mean annual flow.

Rating	Summer low flow (% of mean annual flow)	Description
0 (worst)	<10%	Inadequate to support trout.
1	10-15%	Potential for trout support is sporadic.
2	16-25%	May severely limit trout stock every few years.
3	26-55%	Low flow may occasionally limit trout numbers.
4 (best)	>55%	Low flow may very seldom limit trout.

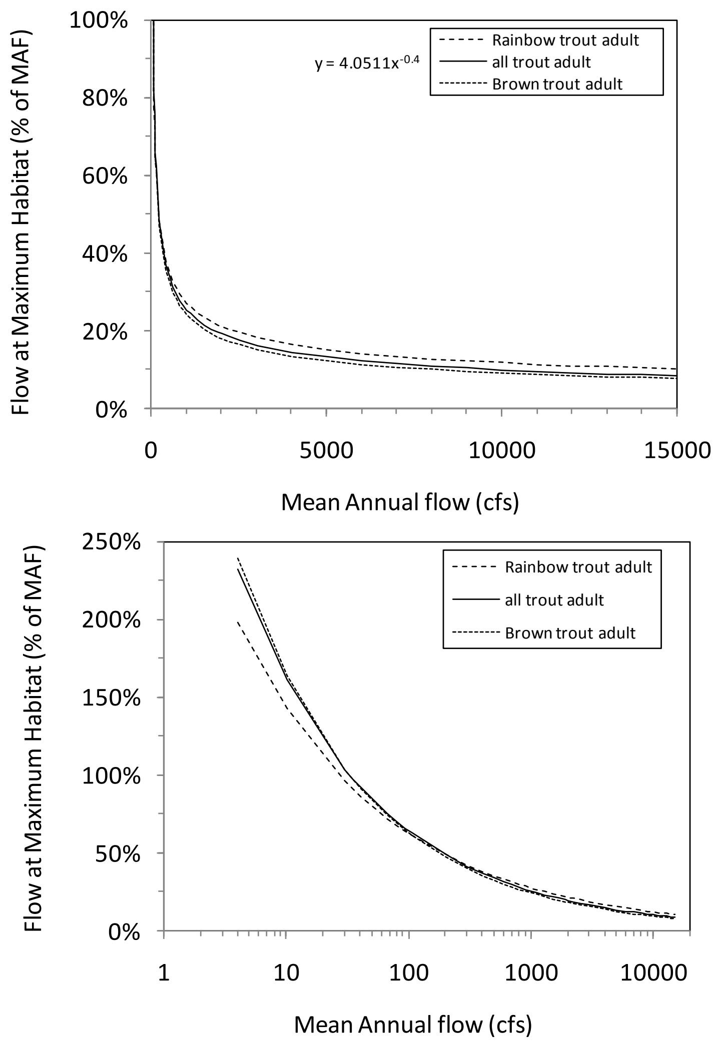


Figure 11 Relationship of flow for maximum trout habitat to mean annual flow (MAF). The same plot is presented on a linear axis (upper plot) and log axis (lower plot) for mean annual flow. Equations were derived from Hatfield and Bruce (2000) for a scenario of latitude 41° (latitude is a second factor in the models for rainbow trout and “all trout”). The equation for “all trout” is presented (see upper-graph), after conversion from m^3/s to cfs.

Warm Water Fishes – Interior Western

Several extensive and long-running monitoring programs have documented fish communities of the Colorado, San Juan, and upper Colorado tributaries. Studies on the San Juan covered too narrow a flow range, documenting a return to a near-natural flow regime (Navaho Dam relicensing investigations). Conversely, studies on the lower Colorado River (below Glen Canyon Dam) lack spatial or temporal reference data for natural conditions (by political decree, see Lovich and Melis 2007). Despite the abundance of data from the lower Colorado River, only investigations from the upper Colorado tributaries provided a wide spectrum of flow conditions, including natural and altered conditions.

Bestgen et al. (2006) present data for a wide range of species over a long period (1962-2006). Spatial coverage is limited (Green River below Flaming Gorge Dam), but this represents one of few studies documenting pre-dam conditions. Reproductive success was reported for fish of the Green River, under various management regimes (temperature and flow manipulation). These data are plotted in response to magnitude of peak flow, low flow and summer temperature (Figure 12). The correlation with both peak flow (positive response) and low flow (negative response) is adequate. However, temperature is the better descriptor of variation in the data (higher R^2), limiting the use of coincidental flow-changes as a causal predictor of fish reproduction.

All 11 species of warm water fish stopped breeding in this reach of the Green River after completion of Flaming Gorge Dam (Bestgen et al. 2006). Outflow was sourced from the cold depths of the reservoir (below the thermocline) which reduced summer temperatures to 6 °C (from 22 °C mid-reach). Seasonal flow variability was also reduced significantly. The installation of variable-depth penstocks in 1978 increased river temperatures to 13 °C (considered optimal for introduced trout), but the flow regime remained stable. The temperature rise alone was adequate for seven species of native fish to start reproducing again, and this period represents the high outlier for the flow response plots in Figure 12. Subsequent operational changes produced a flow regime closer to natural conditions (higher peak flow and reduced low flow), and the number of species reproducing increased to nine (humpback chub and bonytail are yet to recover). But interpreting this latest increase as a response to flow is complicated by the concurrent increase in temperature. Reservoir discharge temperature was increased, and lower flows in summer allow the river to warm more rapidly. Clearly temperature is a critical issue for the persistence of warm water fish in highly regulated rivers, and further investigations were necessary to distinguish the importance of flow.

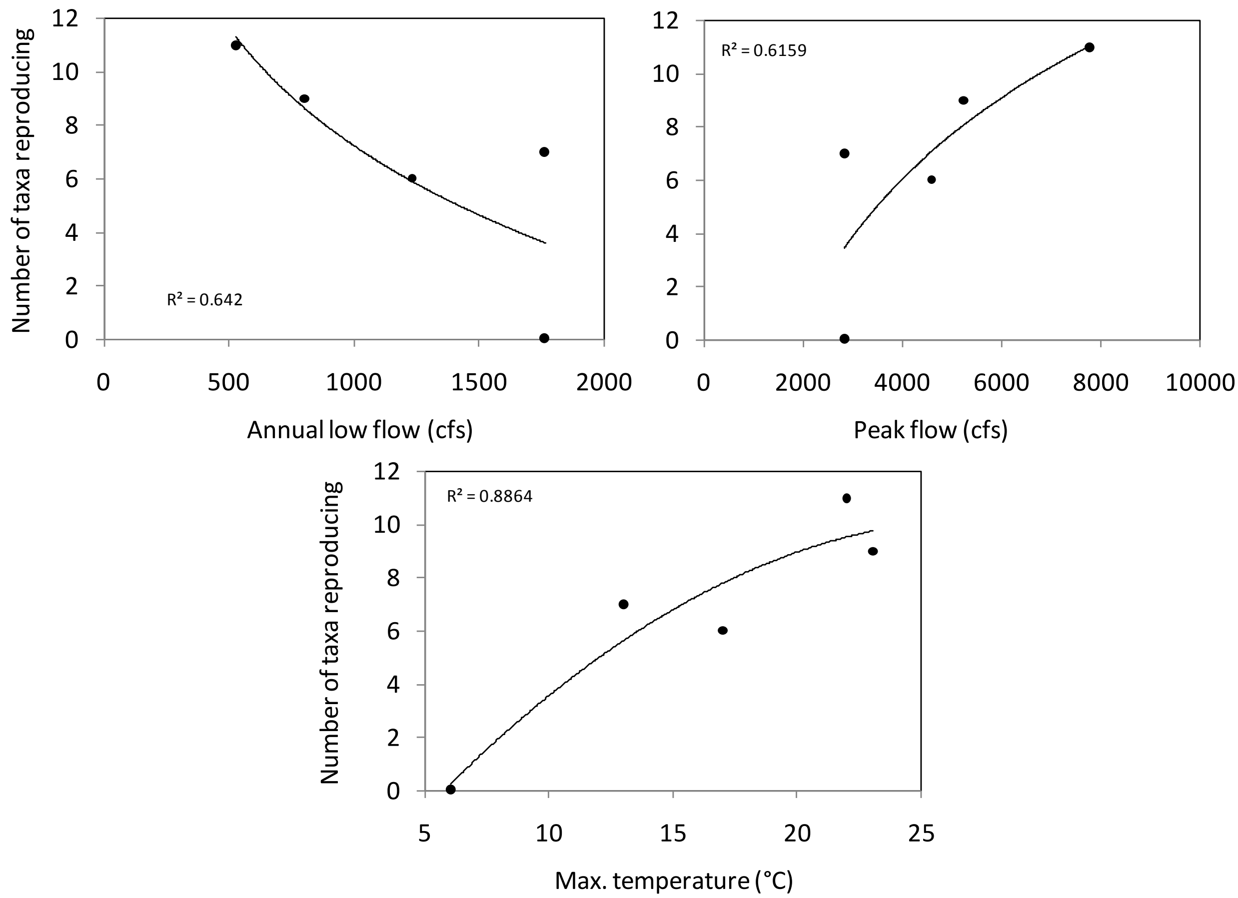


Figure 12 Response of warm water fish to low flow, peak flow and temperature (data from Bestgen et al. 2006). Fish response is measured as the number of taxa reproducing (maximum of 11 taxa including mountain whitefish, humpback chub, bonytail, roundtail chub, Colorado pikeminnow, speckled dace, bluehead sucker, flannelmouth sucker, mountain sucker, razorback sucker and mottled sculpin). Data points represent periods under various dam operations (including pre-dam) on the Green River between Flaming Gorge Dam and Yampa confluence.

By employing data from rivers where temperature is not a major limiting factor, it is possible to distinguish the effects of flow modification. Anderson and Stewart (2007) provide data across a wide range of flow conditions, representing gradients of flow modification, inter-year and site variability, using comparable methods. This study provided recent fish biomass and flow data for the Yampa, upper-Colorado, Gunnison and Dolores Rivers (biomass units are standardized by area fished, which enables comparison between different sized rivers). The four rivers have adequate summer temperatures for warm water fishes, and so provide a better depiction of flow response, when temperature is not an overriding issue. The Gunnison is the most regulated of the four rivers, but the study reaches were far enough downstream of dams for temperatures to exceed 18 °C (daily average) in summer (U.S. Fish and Wildlife data). Anderson and Stewart (2007) provide data for four species of native, large-bodied, warm water fish, including:

- bluehead sucker (*Catostomus discobolus*); feed on benthic algae and invertebrates; rocky riffle habitat (Ptacek et al. 2005).
- flannelmouth sucker (*Catostomus latipinnis*); feed on benthic algae and invertebrates; habitat generalist (Rees et al. 2005a).
- roundtail chub (*Gila robusta*); feed on algae, invertebrates and fish; occupy deep, low-velocity habitats with cover (Rees et al. 2005b); species of special concern.
- Colorado pikeminnow (*Ptychocheilus lucius*); piscivore (fish eater); inhabits deep pools and backwaters, feeding in riffles at night (Modde et al. 1999); federally endangered.

Three species (bluehead sucker, flannelmouth sucker, roundtail chub) demonstrate a positive response to increased low flows (Figures 13 and 14). The logarithmic response represents a steep decline in biomass for flows less than 300 cfs, and a gradual response at higher flows. No zero biomass values were observed above 200 cfs for bluehead sucker, flannelmouth sucker or roundtail chub. Preference for higher flows may reflect improved physical habitat, or increased productivity of food sources (e.g. larger areas of benthic algae). It must be kept in mind that total stream area increases at higher flows, and therefore a constant biomass *per unit area* actually represents a gradual increase in *total* biomass with flow.

The response of Colorado pikeminnow to low flow (summer-autumn) was weak (Figure 14). The biomass of pikeminnow (per unit area) did not appear to benefit from elevated low-flows in regulated rivers, beyond basic persistence. This likely reflects other limiting factors, given that pikeminnow are rare or absent in all three regulated rivers, despite a wide range of low flows. By comparison, the free-flowing Yampa River supported a higher biomass of pikeminnow (per unit area) in years with low flows greater than 30 cfs. Inadequate low flow may have the potential to limit the population, with detailed assessments on the Yampa recommending 93 cfs to maintain habitat for Colorado pikeminnow (Modde et al. 1999) and much higher flows for the Green River (Muth et al. 2000). Low flows that are adequate for bluehead sucker, flannelmouth sucker and roundtail chub may also be adequate for pikeminnow, with no data to suggest otherwise.

Three species (roundtail chub, bluehead and flannelmouth sucker,) show a negative, but weaker, response to specific peak flow compared to the low flow response (Figure 13 and 14). The flow required to disturb or scour a stream bed is a product of, among other things, channel size. This is presumably why specific peak flow (peak flow per unit watershed area) produced a clearer response (higher R^2) for all species, compared to total flow, and is presented here. Specific peak flows that are greater than 2.5 cfs/mile^2 were associated with reduced potential biomass of bluehead sucker, flannelmouth sucker and roundtail chub. The magnitude of low flow is a better predictor of biomass than peak flow for these three species (higher R^2 value and smaller P-value, Figure 13 and 14). Brouder (2001) documented a positive response for juvenile roundtail chub to the magnitude of peak flow, but this did not translate to higher catches of adult chub. Conservation Assessment reports highlight migration barriers and introduced fish as primary threats to these fishes, but research on mechanisms of flow regime effects is limited (Ptacek et al. 2005; Rees et al. 2005a; Rees et al. 2005b).

In contrast to the response of the other three species, Colorado pikeminnow did show a positive correlation with specific peak flow (Figure 14). This federally endangered species was rarely encountered in all rivers but the Yampa, which is the only unregulated river of the four studied. The upper bound (P-value 0.017) therefore describes the response of Yampa River pikeminnow to specific peak flow (all biomass data points >25% are from the “Sevens” site on the Yampa River).

Bestgen et al. (2007a) reported a population decline of Colorado pikeminnow in the Green River basin post-2000 (Yampa sites included in study). The decline was largely attributed to recruitment failure, which has fallen short of adult mortality since the late 1990s. Pikeminnow biomass at the “Sevens” site on the Yampa, which supports the highest biomass of sites monitored by Anderson and Stewart (2007), demonstrated a stronger correlation with peak flow than year (R^2 = 0.813 and 0.247 respectively), indicating that peak flow is not a pseudo correlate for temporal population decline. The response of the Green River basin population to flow is likely to be complex and time lagged, compared to the relationships provided here that describe a more immediate flow response. Given the long life span of Colorado pikeminnow (>6 years to maturity) and distant rearing habitat (mid and lower Green River), the year-to-year variation in biomass described by specific peak flow (Figure 14) is more likely a product of mortality rather than variable recruitment (movement of adults between sites may also influence results). For example, predation by northern pike may increase in the absence of flow disturbance to scour its preferred macrophyte habitat (Bestgen et al. 2007b). Detailed assessments of flow requirements for Colorado pikeminnow are presumably available for critical habitat reaches, including rearing habitat. The relationships derived here will go some way to identifying impoverished flow regimes further afield.

It seems likely that prescribing an upper limit on peak-flows based on the weak negative relationship demonstrated by the other three species (smaller R^2 value and higher P-value, Figure 13 and 14) could be unnecessarily detrimental to Colorado pikeminnow. Notably, Muth et al. (2000) placed no upper limit on their peak flow recommendations for the Green River.

Colorado pikeminnow share traits in common with other federally endangered fishes of Interior Western rivers (bonytail, humpback chub, razorback sucker). Olden et al. (2006) class these fish as long-lived, preferring slow to moderate velocities, and place them in the same reproductive guild (non-guarding, open-substrate). Peak flows appear to be a critical issue for these endangered fish. Muth et al. (2000) stated that recovery of razorback sucker require peak flow of sufficient frequency, magnitude, and duration to inundate floodplain habitats (for the growth and survival of juveniles). The same authors noted that spring flows provide spawning cues and prepare spawning habitat for humpback chub. In the absence of flow-ecology relationships for the full range of fish fauna for the Interior Western rivers, it may be reasonable to assume the positive response of adult pikeminnow to peak flow is typical of other federally endangered species that share similar life history strategies and habitat needs.

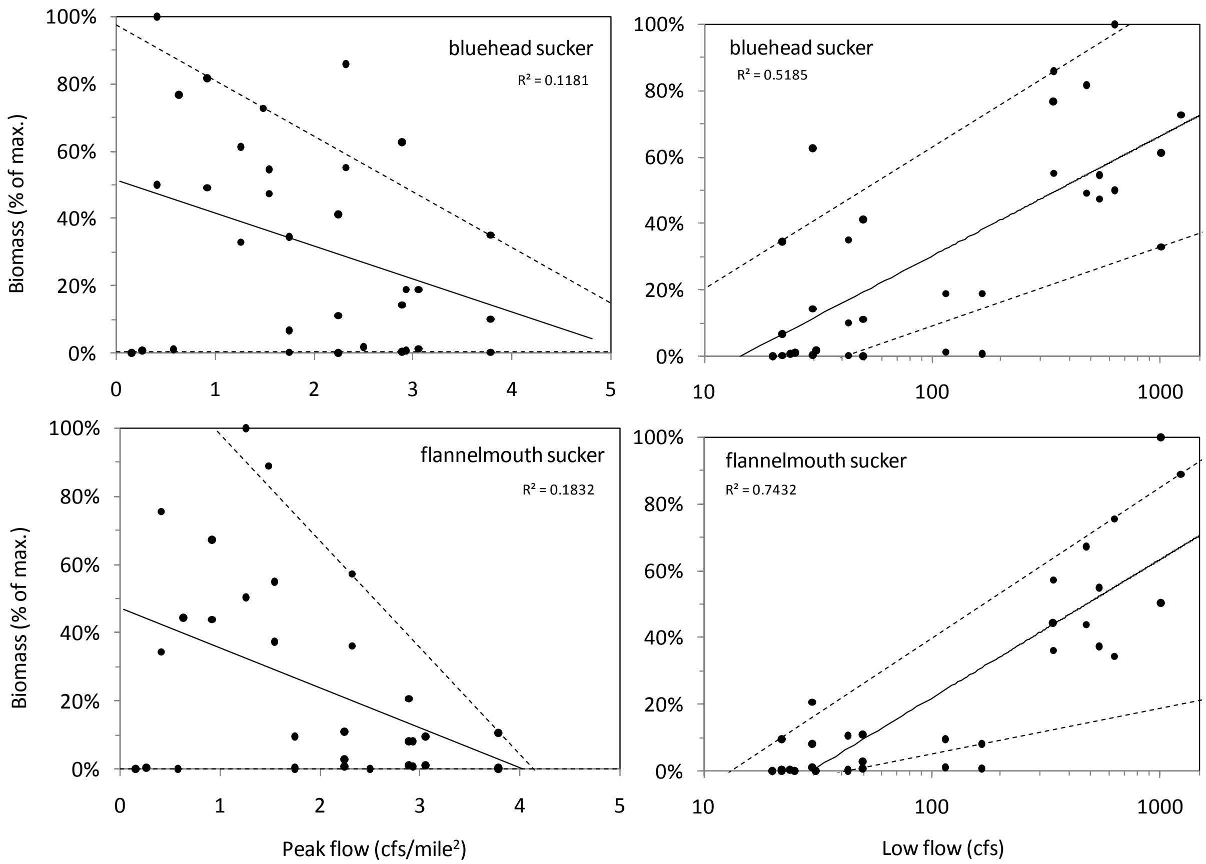


Figure 13 Response of warm water fish (bluehead and flannelmouth suckers) to specific peak flow (left plots) and low flow (right plots, log scale). Data are sourced from Anderson and Stewart (2007). Fish biomass was measured in kilograms per hectare, and is standardized by the observed maximum. Annual peak flow (cubic feet per second, 24-hour average) was divided by watershed area (square miles) to provide a comparable measure of disturbance and inundation for different sized rivers. Low flows are minima for the summer-autumn period (24-hour average), presented on a log scale. Standard regression lines (solid line) and corresponding R^2 values are presented, along with 10% and 90% quantile regression lines (dashed lines). P-values for 90% quantile regressions are, clockwise from top-left, 0.177, 0.054, 0.094 and 0.001.

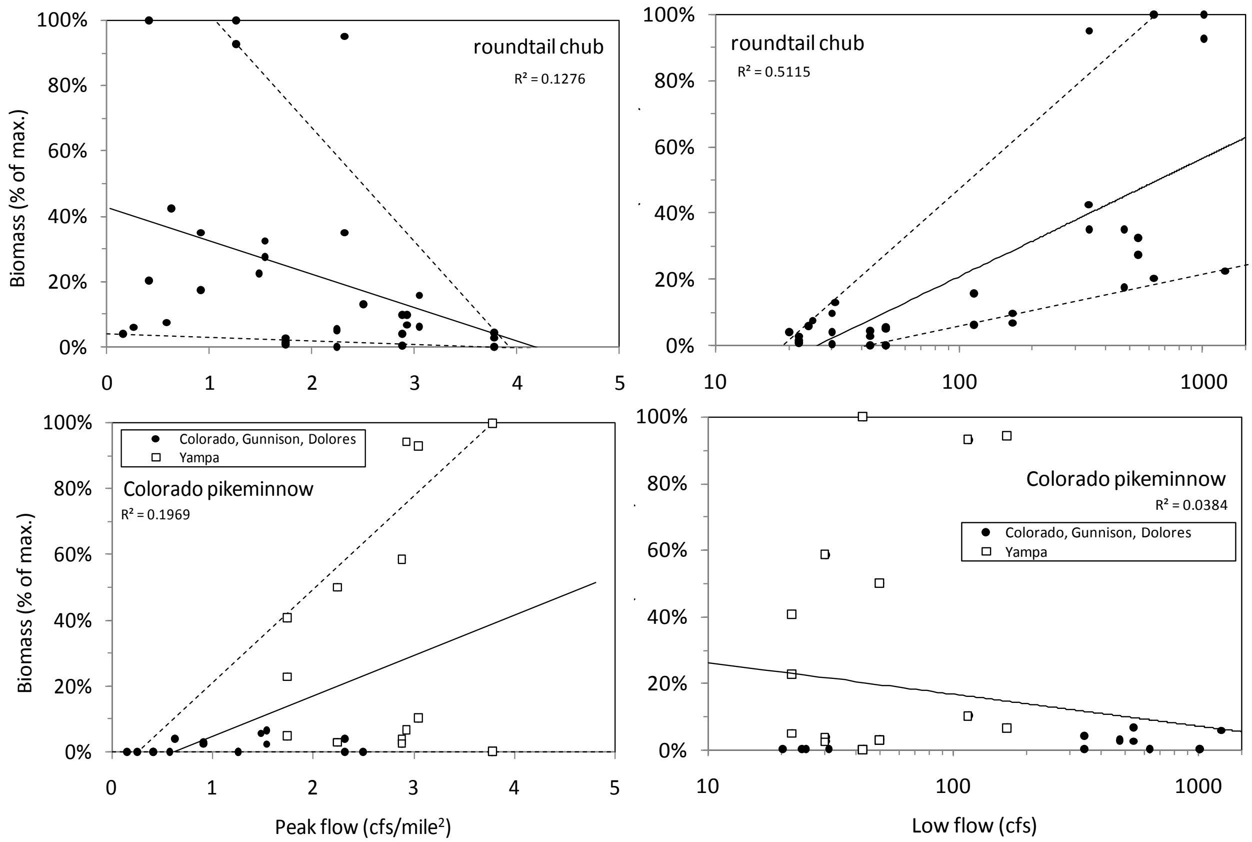


Figure 14 Response of warm water fish (roundtail chub and pikeminnow) to peak flow (left plots) and low flow (right plots). Data are sourced from Anderson and Stewart (2007). Fish biomass was measured in kilograms per hectare, and is standardized by the observed maximum. Annual peak flow (cubic feet per second, 24-hour average) was divided by watershed area (square miles) to provide a comparable measure of disturbance and inundation for different sized rivers. Low flows are for the summer-autumn period (24-hour average), presented on a log scale. Standard regression lines (solid line) and corresponding R^2 values are presented, along with 10% and 90% quantile regression lines (dashed lines). Yampa River results are distinguished for Colorado pikeminnow to highlight their paucity in other rivers (regression based on all 4 rivers). P-values for 90% quantile regressions are, clockwise from top-left, 0.297, 0.008, 0.017 and 0.756 (latter not presented).

The effect of introduced fish is a critical issue for warm water fish of Interior Western rivers. The mechanisms of effect on native fish (e.g. competition, predation, hybridization) potentially have complex interactions with flow and temperature. Generally speaking, the more severe environmental conditions of Interior Western rivers in their natural state (extremes of flow, turbidity and temperature) are expected to favor native fish (Olden et al. 2006). But attempts to restore natural conditions in regulated rivers (excluding sediment regimes) sometimes fail to reduce numbers of introduced fish (Brooks et al. 2000). Flow conditions that provide suitable habitat for native fish are a fundamental starting point for water management in Colorado rivers, and hence are the basis of relationships presented in this report. As the relationships presented here are intended as a screening tool (rather than site-specific flow requirements), we have not attempted to generalize flow provisions that both reduce numbers of introduced fish and provide suitable habitat for native species.

Warm-water fishes - Great Plains

Temporal changes across the Great Plains of Colorado have left half of the fish species (19 out of 38) extirpated (locally extinct), endangered, threatened, or classed as species of special concern (Scheurer et al. 2003a). Altered flow regimes are a critical issue in the decline of these fishes, together with migration barriers that prevent recolonization when flow does return (Fausch and Bestgen 1997). Unfortunately, we lack historical data and reference conditions to evaluate quantitative response to flow for Great Plains fishes.

Lohr and Fausch (1997) identified different fish communities inhabiting mainstem rivers, perennial tributaries and intermittent tributaries of the lower Purgatoire watershed (Table 4). Mainstem rivers that receive snowmelt runoff from Rocky Mountain headwaters (e.g. South Platte, Purgatoire), support different fish communities than smaller tributaries that originate on the plains (Table 4). Tributary streams can be perennial where groundwater levels are adequate, but they are often intermittent. We can define intermittent streams as those with permanent pools that are connected only seasonally by flow. These pools might extend to short flowing sections of stream, revealing the continuum between intermittent and perennial. Habitat preference data from Conklin et al. (1995) at least support the main-river dependence of channel catfish (prefer moderate depths and velocities), and the smaller stream affinity of plains killifish (prefer slow, shallow water). Red shiner and sand shiner do not need particularly fast or deep low flows (Conklin et al. 1995), and their apparent absence from perennial tributaries suggests these are too small to satisfy even modest flow requirements (Table 4). Fausch and Bestgen (1997) observed that larger rivers (e.g. mid- and lower Platte) support more large-bodied fish that live longer than fish characteristic of tributary streams.

Table 4 Fish communities inhabiting different stream types in the Purgatoire River watershed, as observed by Lohr and Fausch (1997). The authors distinguished these communities based on percent occurrence in streams of varying persistence of water (e.g. mainstem river species present at 50% of main river sites, and less frequently elsewhere), and this data is reproduced in the right columns. The lowest flows reported by Lohr and Fausch (1997) or Fausch and Bramblett (1991) are also presented as an approximation for summer low flows. See also Table 6.2 in Fausch and Bestgen (1997) for a broader description of fish communities on the plains.

Community Type	Species	Common name	percent occurrence		
			Mainstem river	Perennial tributary	Intermittent tributary
Mainstem river					
	Cyprinella lutrensis	red shiner	100	0	3
	Platygobio gracilis	flathead chub	100	14	0
	Notropis stramineus	sand shiner	67	0	3
	Rhinichthys cataractae	longnose dace	100	0	0
	Ameiurus melas	black bullhead	92	14	33
	Ictalurus punctatus	channel catfish	92	0	3
Perennial tributary					
	Campostoma anomalum	stoneroller	25	57	27
	Fundulus zebrinus	plains killifish	42	71	17
Generalist (including intermittent)					
	Lepomis cyanellus	green sunfish	42	43	93
	Pimephales promelas	fathead minnow	33	71	40
	Catostomus commersoni	white sucker	83	57	33
	Lowest reported flow	(cfs)	23	0.1	0

Large-bodied species and those with specialized reproductive strategies were often the first to disappear after water resource development on the plains, indicating their sensitivity to flow change. Mean annual flow of the Arikaree River, which originates on the plains, has declined steadily since 1960 and is associated with the loss of five species of fish, including plains minnow, suckermouth minnow, river shiner, stonecat and flathead chub (*pers. comm.* Jeffrey Falke, Colorado State University). Eberle et al. (1993) documented the extirpation of two of the same species (plains minnow and flathead chub) from sections of the Arkansas River, as did Hargett et al. (1999) from the Cimarron River (Kansas). These losses were attributed to the reduced spring/summer peak-flows, which are necessary to carry the neutrally buoyant eggs of plains minnow.

The loss of large river specialists from Great Plains rivers was followed by the decline in small-bodied fish, as flow reductions have continued. Both Arkansas darter and brassy minnow are state threatened, and orangethroat darter is a species of concern. Red shiners and sand shiners were found to benefit from occasional return of seasonal flow by Hargett et al. (1999), and a similar response was observed for Arkansas darter (Labbe and Fausch 2000). The harsh conditions of isolated pools are tolerated by several species, but each depends on flow returning seasonally in order to persist.

Those inhabiting pools of intermittent tributaries are dependent on adequate depths to avoid drying out in summer or freezing to the bottom in winter, as well as depending on wet-season flow to allow reproduction and dispersal (Labbe and Fausch 2000; Lohr and Fausch 1997; Scheurer et al. 2003b). Pool depth depends on groundwater level (Falke and Fausch 2007), and the peak flows that form these channel depressions (Labbe and Fausch 2000). The harsh environment in these pools (high summer temperature, low oxygen, flood disturbance and poor connectivity) is believed to restrict establishment of introduced predators, such as largemouth bass (Lohr and Fausch 1997; Scheurer et al. 2003b).

Adequate data were not found to support a quantitative assessment of relationships with peak flow and low flow. Tennant (1976) developed categories of low flow that provide different levels of habitat maintenance (Table 5). These categories were based on habitat data (wetted width, depth, velocity) for large rivers of the northern Great Plains (Republican, North Platte, Shoshone), several of which had montane headwaters. The categories were intended to represent both cold and warm water fishes, though individual species requirements were not specifically investigated. Only the 10% category is an instantaneous flow, compared to the higher thresholds which Tennant labelled simply as baseflows. This implies the 10% category is tolerable for a shorter duration than the higher thresholds. Application of the higher thresholds to 24-hour mean annual low flow seems appropriate for the purpose of this report, and maintains consistency for application (one could also argue that baseflow refers to a mean monthly flow).

The Tennant method was reviewed by Mann (2006) for its representation of physical habitat (depth, velocity, width and weighted usable area). Mann concluded that Tennant’s categories provide a reasonable representation of low-gradient streams (<1%) such as the Great Plains area (from correlations for Nebraska sites). Tennant’s method is only considered appropriate for mainstem rivers of the Great Plains, given Tennant’s use of large rivers in his study and the underestimation of flow requirements for small streams that is produced by the Tennant method (Jowett 1997; Orth and Leonard 1990).

In addition to low flow categories from Tennant (1976), this method was adapted by Tessmann (1980) to provide month-specific flows for the northern Great Plains (Table 6). The decision criteria effectively place lower and upper limits on the degree of flow modification, and it appears all are based on Tennant’s category for “good habitat” (40%). The original publication was not available for review, and we have no information to interrogate the methods appropriateness for peak flows. But these revised categories may go some way to describe the response of fish habitat to peak flow for mainstem rivers of the Great Plains, in the absence of alternatives.

In addition to mainstem rivers, guidelines are also needed to assess response to flow alteration in tributary streams of the Great Plains. Because of the degree of flow modification, and large proportion of threatened species in remaining habitats, it might be fair to assume that both peak flows and low flows are presently stressed in tributary streams of the Great Plains. The number of extirpations (local extinctions) suggests there is little or no buffer remaining in the system to offset further changes such as global warming or further water abstraction. In the absence of flow ecology relationships, detailed site specific studies may be necessary to determine otherwise.

Table 5 Low flow categories for maintaining various levels of habitat quality, expressed as a percent of mean annual flow. Tennant (1976) developed these categories, which are applicable to mainstem rivers of the Great Plains.

	Low flow (% of mean annual flow)
Optimum habitat	60%-100%
Outstanding habitat	60%
Excellent habitat	50%
Good habitat	40%
Fair or degrading habitat	30%
Poor or minimum habitat	10%
Severe degradation	<10%

Table 6 This expansion of Tennant's categories by Tessmann (1980, as presented in Annear et al. 2004) provides guidelines for minimum monthly flows for maintaining good habitat in mainstem rivers of the Great Plains. MMF = mean monthly flow; MAF = mean annual flow.

	Minimum monthly flow
MMF < 40% of MAF	MMF
MMF > 40% of MAF and, 40% of MMF < 40% of MAF	40% of MAF
40% of MMF > 40% of MAF	40% of MMF

Recreation – Canoeing, Kayaking, Rafting

Rood et al. (2006) investigated flow relationships for recreation involving non-motorized boats. The study focussed on Rocky Mountain rivers of Alberta, Canada, but also assessed rivers further south (including Colorado). Three methods were used to measure recreational flows, including paddler surveys, stage-discharge modelling (for target depths) and expert judgement from guide books. Recreational flow analysis was found to be simpler and less stochastic, compared to instream flow determination for fish or riparian vegetation. They proposed the "Alberta equation" to provide an initial estimate for recreational flows for rivers, especially those draining Rocky Mountain areas (based on mean annual flow). Both the minimum flow and preferred flow for recreation are reproduced in Figure 15. The authors posed two qualifications. First, the equations are believed poorly suited for very large rivers, which seldom have flows that are insufficient for paddling. Second, they considered the equations unsuitable for the most challenging reaches as most paddlers restrict usage of Grade-V white water to low flows.

The equations provided by Rood et al. (2006) provide an excellent basis for desk-top assessments of reaches that are known to have value for recreational paddlers. As with the other guidelines provided in this document, this should not override or replace site-specific investigations where recreational use is likely to be a critical issue.

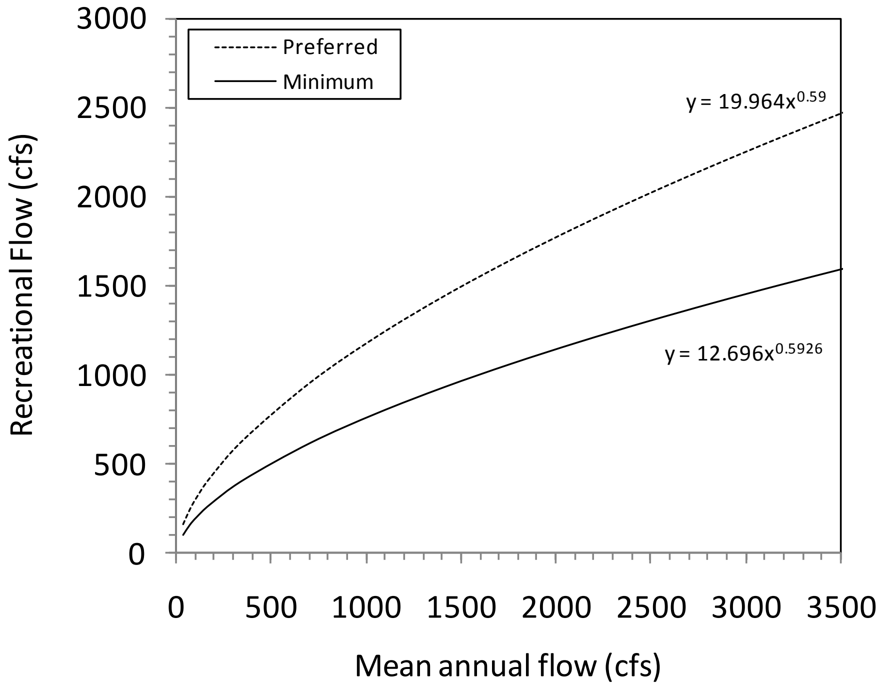


Figure 15 Recreational flows for canoeing, kayaking and rafting (from Rood et al. 2006). Both the minimum flow and preferred flow are presented (converted to cfs from m^3/s units of original publication), and Pearson R^2 values for these regression equations were 0.94 and 0.96 respectively.

4. Summary

This report encompasses an unprecedented range of ecosystems and stream types for the state of Colorado, in an effort to describe their response to flow change. Cottonwoods of the Great Plains, trout of high mountain streams and sucker fish of western canyons are among the plants and animals covered. Drawing on existing scientific research that is intensive and rigorous, but often site-specific, we developed general relationships to improve our understanding of the majority of streams in Colorado.

Complex relationships between ecosystems and their environment are simplified here for the purpose of providing a practical screening tool for water managers. Just as water restrictions have consequences for local economies and our standard of living, basic persistence of stream communities is inextricably linked to flow. The need for food, space to live and successful reproduction often have complex and competing dependence on flow. For example, the same high flow that flushes spawning gravels of silt may wash away newly hatched fish if it occurs at the wrong time of year. The natural flow regime of a river has the best chance of maintaining the plants and animals that naturally occur there. Constructing dams and diversions changes the flow regime and immediately incurs some level of risk that the natural ecosystem will change and species will be lost. But change is not guaranteed, or necessarily bad, because animals and plants can cope with a range of flow conditions. We benefit from water abstraction and regulation of rivers. The question then is how much change can river systems tolerate? Using relationships provided in this report, water managers can evaluate the risk of effects from a given flow change and compare these to aspirations of people in the community.

This report is intended to improve our understanding of the effects of flow change, and is by no means exhaustive. The trade-off between practicality for end users and capturing the complex response of aquatic ecosystems is balanced by limiting the application of the results. Using any one flow-ecology relationship on its own will bias the assessment and omit potentially critical issues from consideration. The relationships are generalizations, and are not intended for prescribing flow requirements (e.g. minimum flows). Instead, they were developed for identifying sites where flow is less likely to be adequate, and for identifying potentially critical issues that warrant site specific investigations. Likewise, this tool alone is unsuitable for the restoration of threatened species, given the wide range of issues, in addition to flow, that require consideration (e.g. water quality, migratory access). The relationships provided here are not a replacement for detailed site-specific studies, but instead are complementary. Relationships developed using intensive site-specific studies should not be rejected in favor of relationships described here for the same issue. For example, relationships between trout and flow are expected to be less accurate than site-specific PHABSIM studies. However, other issues that were not previously evaluated (e.g. riparian vegetation) may warrant evaluation using the relationships provided, to ensure broader consideration of ecosystem response.

Relationships based solely on flow are more robust if applied to sites that are otherwise suitable for the species of interest. Limiting the calculation of flow response to streams that support the species (or supported them prior to flow modification) is therefore recommended. Application will require some knowledge of what aquatic communities are likely to be present at the site of interest. The results section provides more detailed accounts of other critical parameters (other than flow) that commonly constrain response to flow for each stream community (e.g. temperature limitation for Interior Western fish).

This report is not a stand-alone tool. Flow statistics representing both the natural (historic) and the modified condition (e.g. before and after dam construction) are required for site-specific application. Relationships between aquatic ecosystems and flow are the principle tools provided here and they can be used to compare and rank sites. But determining an acceptable level of ecosystem change that triggers further action will be up to end users. Completing an assessment will highlight competing demands among non-consumptive water users (e.g. flow for trout versus riparian vegetation). Developing an understanding among end-users of these competing demands is an important output in its own right. Round tables of interested parties provide an excellent forum for clarifying the specific objectives for flow management in the face of such competing demands.

The user will also need to determine the stream type of the study site in order to select the appropriate relationships (see Stream Types in the Methods section for a description of each). A comprehensive stream classification was beyond the scope of this study, and subsequent development of such a tool would have many advantages for non-consumptive needs assessment in Colorado. We have simplified the diversity of Colorado streams into three classes; Rocky Mountains, Great Plains and Interior Western. These recognizable land forms provide a correlate for major drivers of stream ecosystems (e.g. climate, soil, slope). The rapid transition from mountains to plains simplifies stream classification for Colorado, and flow-ecology relationships are intended to capture variability within each stream type (e.g. moderate to high gradient mountain streams). Exceptions to this are noted in the results (e.g. mainstem versus tributary habitats of the Great Plains). Transition zones do exist between stream types, and the length of these transition zones will vary. Relationships were not developed for transitional reaches, given their implicit variability (transition from one stream type to another over a short distance). Application of relationships intended for both stream types that border the transition zone (assuming each community is present) is expected to provide some appreciation of flow response. For example, sections of Fountain Creek are transitional between the Rocky Mountains and the Great Plains. Here, flow relationships for trout, riparian vegetation and warm-water fish of the Great Plains may all apply. The adequacy of these relationships for transition zones cannot be determined with certainty, so high-value stream communities would warrant detailed assessments.

5. Application

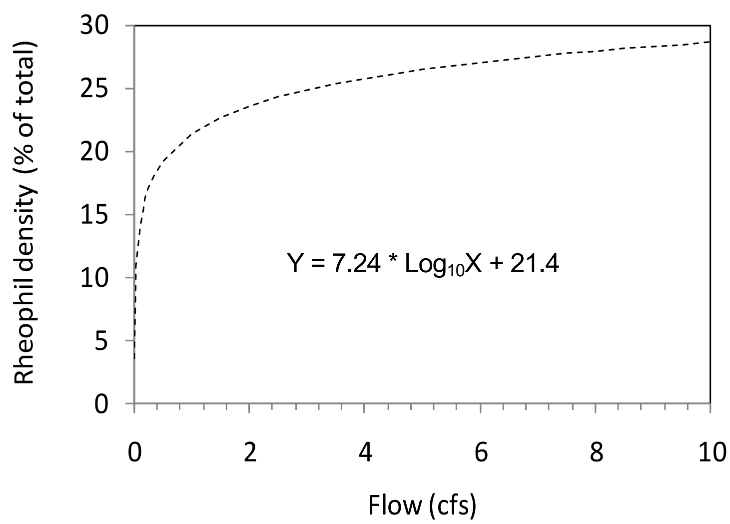
The results section provides details for the derivation of individual flow-ecology relationships. This section summarizes the results to clarify which relationships are most appropriate for use, and to provide a quick reference for implementation. The user must understand the limitations and qualifications of the relationships, as described in the Results and Summary sections. Plots are cross referenced back to the original figure in the Results section, which may appear different because of the change to linear scales here (cf. log scales, etc.). Relationships are presented in turn for each stream type and community.

Rocky Mountain Streams

Invertebrates

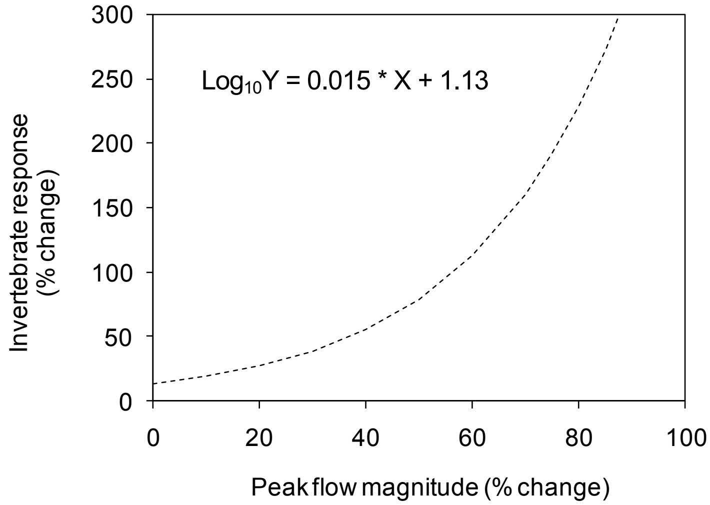
The density of fast-water invertebrates (rheophile species) responds to magnitude of low flow. Method 1 employs the 90% quantile from Figure 6 (Results section). Using the Method 1 equation, calculate a value for the natural flow condition of the site under consideration (24-hour mean annual low flow) and compare this to the value calculated from the existing (altered) flow regime. The response can be presented as a percent change using the two numbers (i.e. [natural – altered] / natural).

Method 1:



Method 2 describes the change of invertebrate populations (diversity and abundance) in response to peak flow alteration. This uses the 75% quantile from Figure 7. Use the percent change in peak flow for the site under consideration (24-hour average annual peak flow) to derive the predicted magnitude of change of invertebrate populations.

Method 2:



Trout

Productive trout fisheries depend on adequate low flow during the summer and autumn. The categories described in Method 3 enable comparison of habitat suitability before and after flow modification, based on summer flows (August-September average) divided by mean annual flow.

Method 3:

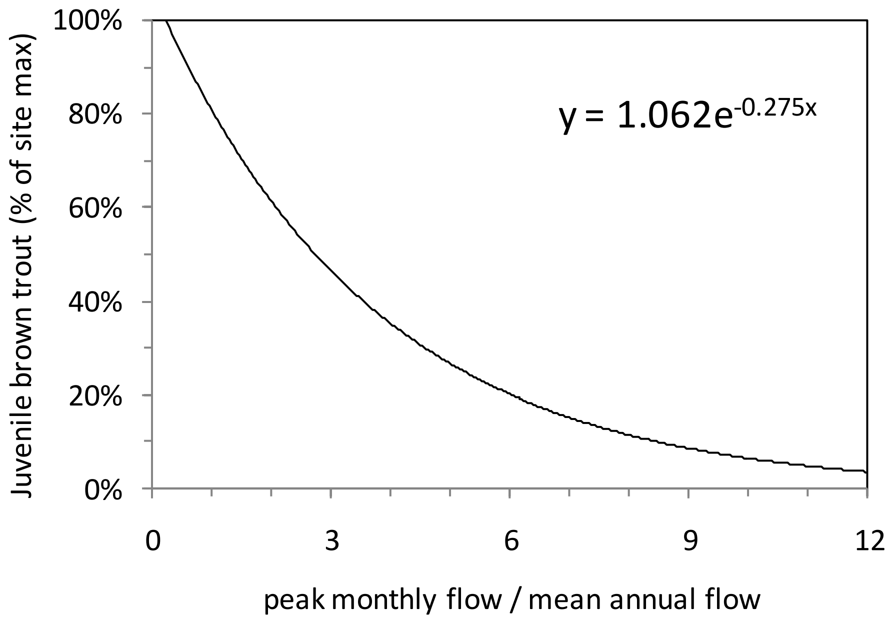
Rating	Summer low flow (% of mean annual flow)	Description
0 (worst)	<10%	Inadequate to support trout.
1	10-15%	Potential for trout support is sporadic.
2	16-25%	May severely limit trout stock every few years.
3	26-55%	Low flow may occasionally limit trout numbers.
4 (best)	>55%	Low flow may very seldom limit trout.

Recruitment of juvenile trout declines in response to peak flow, but elevated peak flows are needed for channel maintenance and excessive trout recruitment can negatively affect the trout fishery (reduced size of adult fish). This effect will be more pronounced for fisheries that presently support high densities of adult trout. The methods provided can be used to evaluate juvenile survival, but not to evaluate overall fishery condition.

Two options are presented for determining recruitment success. As a first option (Method 4), peak monthly mean flows that are <2, <4 and >6 times the mean annual flow indicate high, moderate and poor recruitment of juveniles respectively (derived from Figure 8, Results section). Comparison of peak monthly flows before and after recruitment flow alteration will indicate change in the suitability of peak flows for recruitment. If time series data is available, the inter-year frequency of peak monthly flows for each category will provide a more detailed representation of recruitment success.

Alternatively, the response of trout recruitment to peak monthly flow can be assessed using Method 5. This uses the mean response from Figure 8. Using the Method 5 equation, calculate a recruitment value for the natural flow condition and compare this to the value calculated from the existing (altered) flow regime. The response can be presented as a percent change by dividing the two numbers (i.e. [natural – altered] / natural).

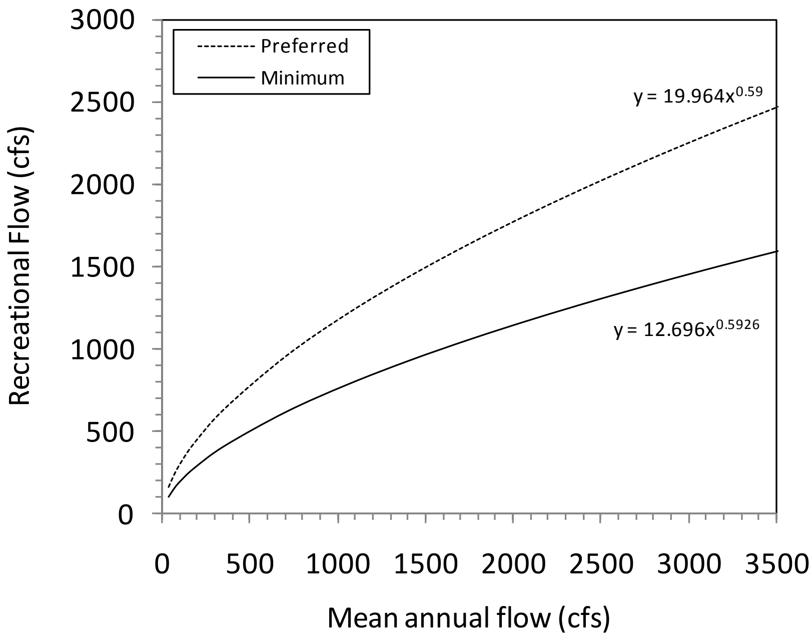
Method 5:



Recreation

Recreational paddlers (rafts, kayaks, etc.) will struggle when flows are too low. Method 6 can be used to estimate a minimum flow and a preferred flow for the site of interest, using the pre-alteration estimate of mean annual flow. Method 6 is reproduced from Figure 15 (Results section). Mean annual low flow for the site of interest can be compared to the minimum flow estimate for paddling, both before and after flow alteration. Preferred flow could be evaluated in the same way or, alternatively, by comparing frequency of days with 24-hour average flows equalling or exceeding the preferred flow before and after flow alteration. The response can be presented as a percent change by dividing the two numbers (i.e. [natural – altered] / natural).

Method 6:



Riparian

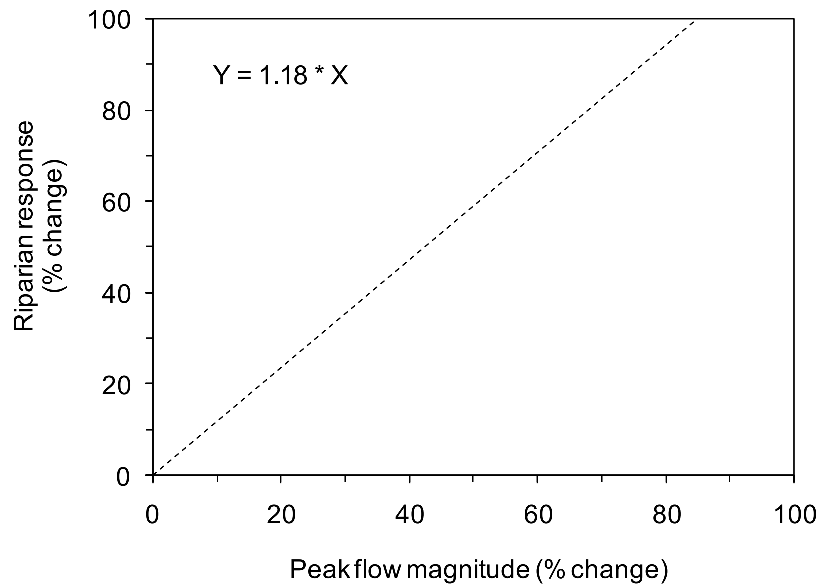
The effects of flow change on riparian communities can be approximated using riparian/peak-flow relationships provided in the Interior Western section (Method 7).

Interior Western Streams

Riparian

Method 7 describes the response of the extent and composition of riparian vegetation to reduced peak-flows. This uses the 90% quantile from Figure 3 (Results section), and can be applied to the site in question using the percent change in peak flow (24-hour mean annual peak flow).

Method 7:



The following table (Method 8) converts the relationship from Method 7 into values for change in peak-flow associated with various levels of change in riparian communities. Such a categorical approach may be useful in some instances (e.g. delineating slight to severe alteration).

Method 8:

Riparian Response	Peak-flow change
10%	8%
25%	21%
50%	42%
75%	64%

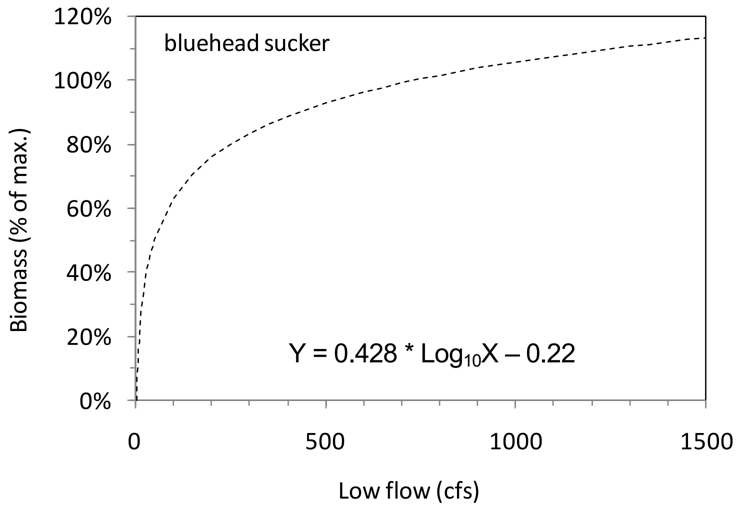
Invertebrates

Reduced peak flows affect invertebrate communities, and the responses described for Rocky Mountain streams (Method 2) is also applicable to Interior Western streams. Adequate data were not available for response to change in low flow.

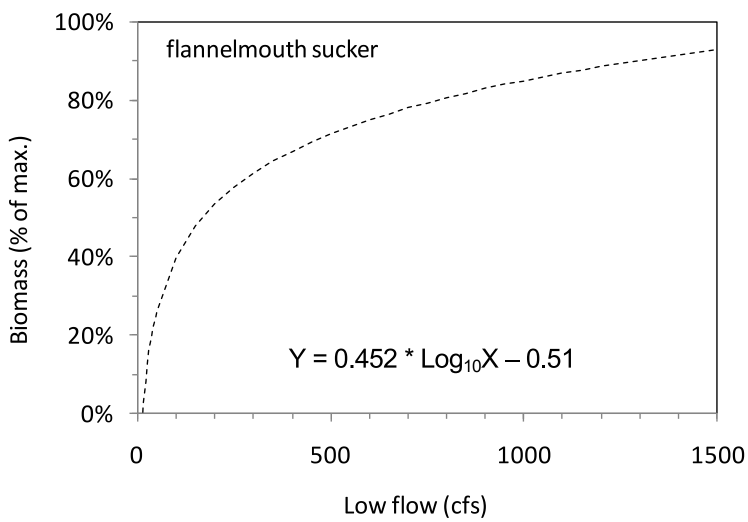
Warm Water Fish

The response of warm water fish to low flow is described in Method 9 for three species inhabiting Interior Western streams. These are based on the 90% quantiles presented in Figures 13 and 14 for low flows. Calculate a value for species relevant to the site under consideration using the 24-hour average low flow for summer autumn. Compare values from flows pre- and post-alteration. The response can be presented as a percent change by dividing the two numbers (i.e. [natural – altered] / natural).

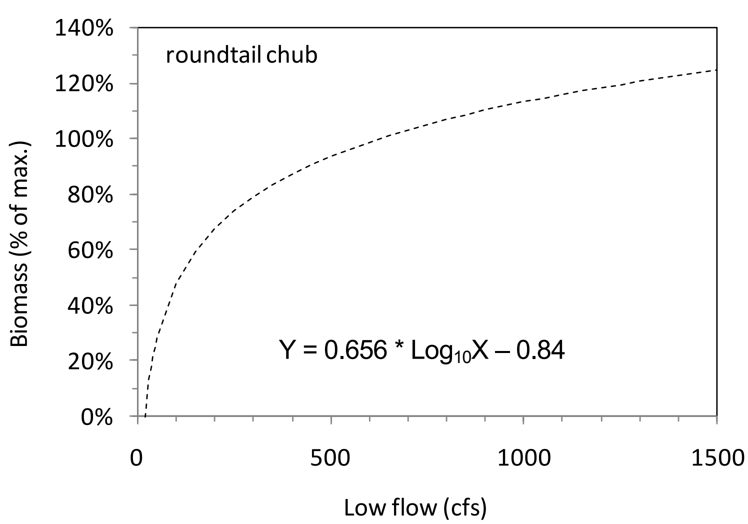
Method 9:



bluehead sucker
Y = 0.428 * Log10X – 0.22
Biomass (% of max.)
120%
100%
80%
60%
40%
20%
0%
0
500
1000
1500
Low flow (cfs)



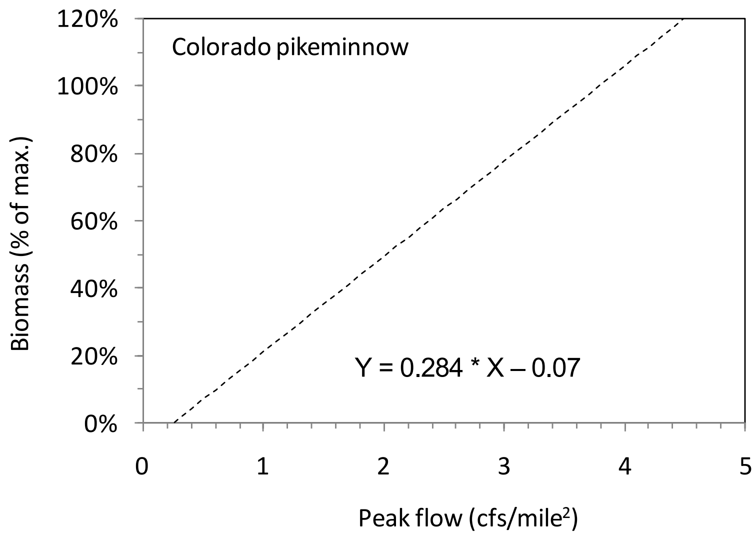
flannelmouth sucker
Y = 0.452 * Log10X – 0.51
Biomass (% of max.)
100%
80%
60%
40%
20%
0%
0
500
1000
1500
Low flow (cfs)



roundtail chub
Y = 0.656 * Log10X – 0.84
Biomass (% of max.)
140%
120%
100%
80%
60%
40%
20%
0%
0
500
1000
1500
Low flow (cfs)

Method 10 describes the response of Colorado pikeminnow to specific peak flow. This is based on the 90% quantile in Figure 14 for peak flow. Pikeminnow are a federally endangered fish that remain in a few Interior Western rivers of Colorado, and peak flow requirements may have been determined already by site specific investigations. In the absence of such information, use Method 10 to calculate a biomass value for the natural flow condition (24-hour average annual peak flow divided by watershed area) and compare this to the value calculated from the existing (altered) flow regime. The response can be presented as a percent change by dividing the two numbers (i.e. [natural – altered] / natural).

Method 10:



Great Plains Streams

Riparian Vegetation

The response of riparian vegetation to altered peak-flow for streams of the Great Plains is described in the Interior Western section using Method 7.

Warm Water Fish

Flow alteration of many tributary streams (streams without Rocky Mountain headwaters) is severe enough to have already eliminated some fish species. Data was not found to support quantitative relationships with flow for these tributary streams. Likewise, limited information was available to quantify the response of fish to change in peak flow for all rivers of the Great Plains (see Results section for options).

For larger mainstem rivers of the Great Plains that have Rocky Mountain headwaters, Method 11 provides categories of response to low flow for fish habitat. This is reproduced from Table 5 in the Results section (Tennant method). These categories can be compared to 24-hour mean annual low flow prior to and after flow alteration for a given site. This will support conclusions on the degree of fish habitat alteration (e.g. good to poor).

Method 11:

	Low flow (% of mean annual flow)
Optimum habitat	60%-100%
Outstanding habitat	60%
Excellent habitat	50%
Good habitat	40%
Fair or degrading habitat	30%
Poor or minimum habitat	10%
Severe degradation	<10%

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Appendix C

The Ecological Limits of Hydrologic Alteration (ELOHA): A New Framework for Developing Regional Environmental Flow Standards

The Ecological Limits of Hydrologic Alteration (ELOHA): A New Framework for Developing Regional Environmental Flow Standards

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Summary

1. The flow regime is a primary determinant of the structure and function of aquatic and riparian ecosystems for streams and rivers. Hydrologic alteration has impaired riverine ecosystems on a global scale, and the pace and intensity of human development greatly exceeds the ability of scientists to assess the effects on a river-by-river basis. Current scientific understanding of hydrologic controls on riverine ecosystems and experience gained from individual river studies support development of environmental flow standards at the regional scale.

2. This paper presents a consensus view from a group of international scientists on a new framework for assessing environmental flow needs for many streams and rivers simultaneously in order to foster development and implementation of environmental flow standards at the regional scale. This framework, the Ecological Limits of Hydrologic Alteration (ELOHA), is a synthesis of a number of existing hydrologic techniques and environmental flow methods that are currently being used to various degrees and that can support comprehensive regional flow management. The flexible approach allows scientists, water-resource managers and stakeholders to analyze and synthesize available scientific information into ecologically-based and socially-acceptable goals and standards for management of environmental flows.

3. The ELOHA framework includes the synthesis of existing hydrologic and ecological databases from many rivers within a region in order to develop relationships between flow alteration and ecological responses. These relationships serve as the basis for the societally-driven process of developing regional flow standards. This is to be achieved by first using hydrologic modeling to build a 'hydrologic foundation' of baseline and current hydrographs for stream and river segments throughout a region. Second, using a set of ecologically-relevant flow variables, river segments are classified into a few distinctive flow regime types that are expected to have different ecological characteristics. These river types can be further subclassified according to important geomorphic features that define physical habitat features. Third, the deviation of current-condition flows from baseline-condition flow is determined. Fourth, flow alteration - ecological response relationships are developed for each river type, based on a combination of existing hydroecological literature and expert knowledge.

4. Lack of precision in the relationships between flow alteration and ecological responses is expected, in part because of the confounding of hydrologic alteration with other important environmental determinants of river ecosystem condition (e.g., temperature). Application of the ELOHA framework should therefore occur in a consensus context where stakeholders and decision-makers explicitly evaluate acceptable risk as a balance between the perceived value of the ecological goals, the economic costs involved, and the scientific uncertainties in functional relationships between ecological responses and flow alteration.

5. The ELOHA framework also should proceed in an adaptive management context, where collection of monitoring data or targeted field sampling data allows for testing of the proposed flow alteration – ecological response relationships. This empirical validation process allows for a fine-tuning of environmental flow management targets. The ELOHA framework could greatly accelerate comprehensive management of river flows to support sustainable goods and services, biodiversity, and human well being in the face of growing human demands for fresh waters on a global scale.

Key Words
Environmental flows, Hydrologic alteration, Streamflow classification, Hydrologic modeling, Hydroecology, River management, Risk-based decision making

Introduction

Water managers the world over are increasingly challenged to provide reliable and affordable water supplies to growing human populations. At the same time, local communities are expressing concern that water development should not degrade freshwater ecosystems or disrupt valued ecosystem services, such as the provision of fish and other sources of food and fiber as well as places for recreation, tourism, and other cultural activities (Postel & Carpenter, 1997; Naiman et al., 2002; Dyson *et al.*, 2003; Postel & Richter, 2003). Aquatic ecosystems support our livelihoods, life styles and ethical values (Acreman, 2001). While people need water directly for drinking, growing food and supporting industry, water for ecosystems often indirectly equates to water for people (Acreman, 1998). There is a fundamental need to address ecological requirements and optimize social well-being across a broad array of water needs to attain sustainability in the management and allocation of water (Gleick, 2003; *Millennium Ecosystem Assessment*, 2003, 2005). Deliberate and strategic design of resilient ecosystems, including freshwaters, is now recognized as a major social-scientific challenge of the 21st Century (Palmer *et al.*, 2004).

Environmental flows is the term applied to explicit management of water flows through freshwater ecosystems such as streams, rivers, wetlands, estuaries and coastal zones to provide an appropriate volume and timing of water flow to sustain key environmental processes and ecosystem services valued by local communities. It is now widely accepted that a naturally-variable regime of flow, rather than just a minimum low flow, is required to sustain freshwater ecosystems (Poff *et al.* 1997; Bunn & Arthington 2002; Postel & Richter 2003; Annear *et al.* 2004; Biggs, Nikora & Snelder, 2005), and this understanding has contributed to the implementation of environmental flow management on thousands of river kilometers worldwide (Postel & Richter, 2003). Despite this tangible progress, millions of kilometers of river and thousands of hectares of wetlands (and the human livelihoods dependent upon them) remain unprotected from the threat of over-allocation of water to offstream uses or to other alterations of the natural flow regime. These threats will only continue to increase with projected growth in the human population and its associated demand for energy, irrigated food production and industrial use (CAWMA 2007), and with uncertainties associated with climate change (Vörösmarty *et al.*, 2000; Dudgeon *et al.*, 2006; Palmer *et al.*, 2008). As water development plans are being formulated to provide greater water security and other social benefits, it will be critically important to ensure that the considerable socioeconomic benefits already provided by healthy freshwater ecosystems are not lost and that those degraded ecosystems be restored.

A sense of urgency has arisen for the need to develop ecological goals and management standards that can be applied globally to streams and rivers across a spectrum of ecological, social, political and governance contexts, regardless of the current stage of water resource development. The imperative to incorporate ecosystem needs for fresh water into basin-wide and regional water resources planning is increasingly recognized at national and international scales (Dyson*, et al.*, 2003; GWSP, 2005; NSTC, 2004; CAWMA, 2007; Brisbane Declaration, 2007 [URL: www.riversymposium.com/index.php?element=2007BrisbaneDeclaration241007]). Unfortunately, the pace and intensity of flow alteration in the world's rivers greatly exceeds the ability of scientists to assess the effects on a river-by-river basis – this despite notable scientific progress in the last decade in developing environmental flow methods for river-specific applications (Tharme, 2003; Annear *et al.*, 2004; Arthington *et al*. 2004; King and Brown, 2006). Thus, a key challenge in securing freshwater ecosystem sustainability is synthesizing the knowledge and experience gained from individual case studies into a scientific framework that supports and guides the development of environmental flow standards at the *regional* scale (Poff *et al*., 2003; Arthington *et al*., 2006), i.e., for states, provinces, large river basins, or even entire countries. Defining environmental flow standards for many rivers simultaneously, including those for which little hydrologic or ecological information presently exists, is necessary for water managers to effectively integrate human and ecosystem water needs in a timely and comprehensive manner (Arthington *et al*., 2006).

In this paper, we present a consensus view from a group of international scientists on a new framework for assessing environmental flow needs that we believe can form the basis for developing and implementing environmental flow standards at the regional scale. This consensus reflects our experiences and knowledge of the science of environmental flows gained through both scientific research and practical applications. We refer to this framework as the "Ecological Limits of Hydrologic Alteration," or ELOHA. Our goal is to present a logical approach that flexibly allows scientists, water-resource managers and other stakeholders to analyze and synthesize available scientific information into coherent, ecologically-based and socially-acceptable goals and standards for management of environmental flows. This presentation of the ELOHA framework focuses primarily on the scientific approaches and challenges of providing the best possible information regarding the range of ecological consequences that will result from different levels of flow modification at a regional scale. We deliberately provide only cursory treatment of the social and policy challenges inherent in gaining adoption of water management goals and implementation of environmental flow standards consistent with those goals. We expect that other authors with expertise in water policy and the social sciences will offer their perspectives on the need for, and challenges associated with, effectively implementing the ELOHA framework in a variety of social and governance contexts.

Historical Scientific Foundations of the ELOHA Framework

The protocol for regional environmental flow assessment described in this paper is grounded in several recent and important scientific advances. First, research over the last few decades has amply demonstrated that ecological and evolutionary processes in river ecosystems are heavily influenced by many facets of a dynamic, historical flow regime (reviewed in Poff *et al*., 1997; Bunn & Arthington, 2002, Lytle & Poff, 2004). Indeed, streamflow has been called the "master variable" (Power *et al*., 1995), or the "maestro … that orchestrates pattern and process in rivers" (Walker, Sheldon & Puckridge, 1995). Much evidence also exists that modifications of streamflow induce ecological alterations (reviewed in Bunn & Arthington, 2002; Poff & Zimmerman, this volume). Thus, both ecological theory and abundant evidence of ecological degradation in flow-altered rivers support the need for environmental flow management. Certainly, other environmental factors besides streamflow (including temperature, water quality, sediment, and invasive species), also regulate riverine ecosystem structure and function, as has been well recognized (e.g., Poff *et al*., 1997; Baron *et al*., 2002; Dudgeon *et al*., 2006). A fuller accounting of the interactions between flow and these other environmental features remains a challenge for advancing the science of environmental flows (and this is discussed more fully below); however, we argue that our present scientific understanding of the role of flow alteration in modifying ecological processes justifies the development of regional flow standards to underpin river restoration and conservation. At a minimum, as society struggles to conserve and restore freshwater ecosystems, flow management is needed to ensure that existing ecological conditions do not further decline (Palmer *et al*., 2005).

A second scientific foundation supporting ELOHA is the extensive development and application of environmental flow methods globally (see Tharme, 2003; Acreman & Dunbar, 2004). These methods, along with the development of hundreds of ecologically-relevant flow metrics and techniques for quantifying human-caused flow and ecological alteration (Richter *et al*., 1996; Puckridge *et al*. 1998; Olden & Poff, 2003; Arthington *et al*., 2004, 2007; Kennen *et al*., 2007; Mathews & Richter, 2007), provide a rich toolbox for environmental flow science. Many of these methods and tools can be directly applied or readily adapted for use in regional environmental flow assessment.

Third, the conceptual foundation now exists to facilitate regional environmental flow assessments. By classifying rivers according to ecologically-meaningful streamflow characteristics (e.g., Poff & Ward, 1989; Harris *et al.*, 2000; Henriksen *et al*., 2006), groups of similar rivers can be identified, such that within a grouping or type of river there is a *range* of hydrologic and ecological variation that can be considered the natural variability for that type. Arthington *et al*. (2006) argued that empirical relationships describing ecological responses to flow regime alteration within river flow types should form the basis of flow management for both river ecosystem protection (proactive flow management) and sustainable restoration (reactive flow management). This perspective represents a major advance by bridging the gap between the simplistic and often arbitrary hydrologic "rules of thumb" presently being used for regional-scale estimation of environmental flow needs and, at the other extreme, the detailed and often expensive environmental flow assessments being applied on a river-by-river basis.

Fourth, developing and implementing environmental flow standards at regional scales ultimately requires employing hydrologic models that can provide reasonably accurate estimates of ecologically-meaningful streamflows in rivers or river segments distributed throughout a region, including those lacking streamflow gauging records (e.g., Snelder & Biggs, 2005; Kennen *et al*. 2008). Hydrologic models can be used to evaluate the nature and degree of hydrologic alteration resulting from human activities and to anticipate the degree to which proposed human activities may further alter the hydrologic regime. With modeled hydrographs, all river segments can be classified hydrologically and ecological information collected from ungauged locations can be used to support the development of relationships between flow alteration and ecological degradation.

Finally, contemporary scientific understanding acknowledges that river management involves complex, coupled social-ecological systems (Rogers, 2006), and if science is to contribute to sustainable water and ecosystem management, it must become engaged in collaborative processes with managers and other stakeholders to illustrate alternative river visions and to help define pathways to achieve socially-desirable goals (Poff *et al*., 2003). The complexity of river systems generates uncertainty in their response to many types of management actions (including flow manipulation); therefore, scientists must be willing to articulate an adaptive learning cycle that uses the best available science to set ecosystem management goals and then uses monitoring to improve understanding of ecological responses to management actions. Ultimately, this approach will allow future management actions to be fine-tuned (Arthington & Pusey, 2003; King, Brown & Sabet, 2003; Richter *et al.*, 2006; Rogers, 2006).

We present the ELOHA framework as a synthesis of a number of existing hydrologic techniques and environmental flow methods that are currently being used to various degrees and that can support comprehensive regional flow management. Many of the basic elements of the framework presented here are now being implemented in a variety of geographical settings and political jurisdictions around the world. As products and summaries of these early ELOHA applications become available, and pertinent tools and techniques useful in ELOHA are described in greater detail, they will be posted on the internet at www.nature.org/ELOHA.

The Scientific Process in the ELOHA Framework

The ELOHA framework involves a number of inter-connected steps, feedback loops, and iterations (Fig. 1). Relationships between flow alteration and ecological characteristics for different river types comprise the key element that links the hydrologic, ecological, and social aspects of environmental flow assessment. These relationships are based on paired streamflow and ecological data from throughout the region of interest. Our description of the ELOHA framework is presented in stepwise fashion, recognizing that various scientific and social processes will likely proceed simultaneously and many need to be repeated iteratively.

The scientific process consists of four major steps, each with a number of technical components, building upon the approach recommended in Arthington *et al*. (2006). It is the express intent of the architects of the ELOHA framework to provide considerable flexibility in the selection of particular input data, tools or analytical methods for accomplishing each step. A risk-based approach is encouraged, which involves choosing the most appropriate model through a trade-off between avoiding the unnecessary expense and effort of developing highly detailed and data hungry models (often applicable at site-specific scales), while generating information and products containing sufficient certainty to provide sufficienct confidence to support decisions at broad regional scalesconfidence (Acremen & Dunbar, 2004; Acreman *et al.*, 2006). Such a risk-based approach may be initiated in many regions by investing in simple tools and using readily-available data, then moving to more complex and expensive approaches, including additional data collection as the need for prediction resolution increases.

1. Building a Hydrologic Foundation

A key feature of the ELOHA framework is a hydrologic database that describes flow regimes not just in "traditional" anthropocentric terms, such as average yield or reliability, but also in terms known to be linked to ecological outcomes (described below). Hydrologic modeling is used to create the hydrographs that form the "hydrologic foundation," which consists of two comprehensive databases of daily (or possibly longer time steps such as weekly or monthly) flow time series representing simulated baseline and developed conditions throughout the region during a common time period. Baseline conditions refer to minimally-altered or best-available conditions (the "reference-site approach," *sensu* Stoddard *et al.*, 2006), whereas developed conditions refer to altered flow regimes associated with both the direct (e.g., water resource development) and indirect (e.g., land use change) effects of human activities.

The hydrologic foundation serves several important purposes. First, it facilitates the use of ecological information collected throughout the region, thereby expanding the number of sites that can be used in developing flow alteration-ecological response relationships beyond only those sites having streamflow gauges. Second, it provides a basis for comparing present-day flow regimes to baseline conditions, i.e., those that served as the template for recent evolution of native species and for shaping ecosystem processes, as well as sociocultural dependencies upon those ecological conditions and processes. Third, it enhances the ability of water managers and planners to understand the cumulative impacts of hydrologic alteration that have already taken place across the region, so that those alterations can be linked to observed changes in ecological conditions and ecosystem services as a basis for forecasting future ecological change in the context of regional water management planning. In a similar vein, the foundation can be combined with other regional environmental information (e.g., non-point pollution sources on agricultural lands) to generate landscape characterizations of management interest.

The coupled baseline and developed hydrologic time series comprising the hydrologic foundation should be developed for all locations in the region where water management decisions, including environmental flow protection, are needed or anticipated. These "analysis nodes" should be identified in close collaboration with water managers who will use the hydrologic foundation to understand and manage water allocation and environmental flows. The baseline and developed-condition hydrographs are the basis for measuring hydrologic alterations, which serve as independent variables in developing flow alteration-ecological response relationships (described in Step 4 below). Therefore, analysis nodes should also be established for all sites at which ecological data to be used in flow alteration-ecological response relationships have been collected or are likely to be collected (Step 3), and they should include the range of geomorphic features at the river segment scale that mediate how habitat availability and diversity are expressed for a given flow regime (see Step 2 below). All of this information should be stored in a relational database and imported into a Geographic Information System (GIS) to enable users to easily access hydrographs and associated flow statistics.

Fig. 2 illustrates the general approach for building the regional hydrologic foundation, (www.nature.org/ELOHA offers several case studies). Briefly, the approach uses records from existing streamflow gauges for a selected time period. The gauge records are segregated into those that represent baseline conditions and those that represent developed conditions. Regression and/or simulation models (e.g., watershed, rainfall/runoff) are used as necessary to estimate streamflows for baseline and developed conditions at ungauged analysis nodes and for time periods not represented in the period of record. For rapidly changing land uses (e.g., urbanization), developed-condition hydrographs could be modeled for both existing and alternative future scenarios, including projected climatic regimes. Ideally, daily streamflows will be generated for the hydrologic foundation, as daily data provide appropriate temporal resolution for understanding most ecological responses to flow alteration. However, in cases where daily data cannot be satisfactorily modeled, a coarser grain of resolution such as weekly or monthly hydrographs can provide some ecologically relevant information (see Poff, 1996) and may serve as a starting point for classification.

Given limited availability of streamflow gauging records with which to calibrate estimates of baseline or developed conditions, and given that climate and river runoff vary naturally over annual to decadal time scales (Lins and Slack, 1999; McCabe and Wolock, 2002), it is desirable to adopt a single time period as a climatic reference period for which baseline and developed-condition streamflows are synthesized and modeled. By using a common climatic reference period for each of these two scenarios, human influences on flow regimes can be separated from climatic influences.

The basic data required to develop the hydrologic foundation are now available for most parts of the globe (Kite, 2000), enabling hydrologists to generate a first-cut approximation of the hydrologic foundation in most, if not all, regions. Prediction accuracy is a significant concern, especially in sparsely gauged regions, but improvements in *a priori* parameter estimation based on remotely sensed land-surface characteristics and the development of Bayesian Monte Carlo techniques have significantly improved the accuracy of hydrologic models (Duan *et al.*, 2006; Schaake *et al.*, 2006). Since the objective of ELOHA is to identify ecologically significant differences in flow regimes between baseline and developed conditions, it is important to quantify apparent differences that arise due to poor model performance and true differences due to water or catchment management. For example, Acreman (2007) distinguished model error from true differences between natural flows and impacted flows downstream of dams in the process of defining ecologically significant thresholds of flow alteration for the European Water Framework Directive in the United Kingdom.

2. Classifying Rivers According to Flow Regimes and Geomorphic Features

River classification is a statistical process of stratifying natural variation in measured characteristics among a population of streams and rivers to delineate river types that are similar in terms of hydrologic and other environmental characteristics. The classification can be developed within any "region" of interest, from those defined by political boundaries to those representing natural biophysical domains, such as physiographic provinces or ecoregions.

River classification serves two important purposes in the ELOHA framework. First, by assigning rivers or river segments to a particular type, relationships between ecological metrics and flow alteration can be developed for an entire river type based on data obtained from a limited set of rivers of that type within the region (Arthington, *et al.*, 2006; Poff, *et al.*, 2006b). For each river type there is a range of natural hydrologic variation that regulates characteristic ecological processes and habitat characteristics (Arthington, *et al.*, 2006; Lytle & Poff, 2004), and that represents the baseline or reference condition against which ecological responses to alteration are measured across multiple river segments falling along a gradient of hydrologic alteration.

Second, combining the regional hydrologic modeling with a river typology facilitates efficient biological monitoring and research design. Specifically, it is possible to strategically place monitoring sites throughout a region to capture the range of ecological responses across a gradient of hydrologic alteration for different river types. This is particularly valuable in regions with sparse pre-existing biological data or where monitoring and research resources are limited.

Hydrologic Classification. In the ELOHA framework, river classification focuses primarily on the hydrologic regime as the main ecological driver. Examples of river types in the United States include stable groundwater fed rivers; seasonally predictable snowmelt rivers; intermittent, rain-fed prairie and desert rivers; and highly dynamic, unpredictable rain-fed perennial rivers (e.g., see Poff, 1996). We recommend classifying rivers according to similarity in hydrologic regime, using flow statistics computed from the baseline hydrographs developed in Step 1. A large suite of flow statistics can be calculated using software packages such as the Indicators of Hydrologic Alteration (Richter *et al.*, 1996), the Hydrologic Assessment Tool (HAT) within the Hydroecological Integrity Process (Henriksen *et al.*, 2006), the River Analysis Package (www.toolkit.net.au/rap), or GeoTools (http://www.engr.colostate.edu/~bbledsoe/GeoTool/). The number of river types in a region should generally reflect the region's heterogeneity in climate and surficial geology, with diverse regions having more river types. Deciding how many river types are appropriate requires a tradeoff between detail (i.e., large number of types with small within-type variability) and interpretability (i.e., small number with small within-type variability). In order to be practical to management, a relatively small number of river types should be defined that capture the major dimensions of streamflow variability. For example, Kennen et al. (2007) defined four river types for the State of New Jersey (ca. 22,000 km^2), and Henriksen et al. (2008) defined three primary and five secondary stream types in Missouri (ca. 178,000 km^2) using a similar approach.. Poff & Ward (1989) and Poff (1996) identified 10-12 river types for the continental United States, but those were subsequently reduced to six general types by Olden & Poff (2003). Further stratifying river types by major environmental features, such as geomorphic setting (see below), will increase the number of relevant river types. For example, the State of Michigan (ca. 248,000 km^2) has identified 11 river types based on a combination of hydrologic regime and temperature conditions (Michigan Groundwater Conservation Advisory Council, 2007). Acreman *et al.* (2008) classified rivers in the United Kingdom into 10 ecological types based on physical basin characteristics to define national environmental flow standards to implement the EU Water Framework Directive. Snelder & Biggs (2002) identified 4 major types of flow sources that were further stratified by other environmental variables to develop a national river segment classification for New Zealand.

Three primary criteria should be considered in selecting a suite of flow statistics for building a river classification. First, if possible, flow metrics should collectively describe the full range of natural hydrologic variability, including the magnitude, frequency, duration, timing, and rate of change of flow events (Poff *et al.*, 1997; Richter *et al.*, 1996; Olden & Poff, 2003; Kennen *et al.*, 2007; Mathews & Richter, 2007). Second, metrics must be "ecologically relevant," i.e., they are known to have, or can reliably be extrapolated from ecological principles to have, some demonstrated or measurable ecological influence (Arthington *et al.*, 2006, Monk *et al.*, 2007) and hence will be important in assessing ecological responses to hydrologic alteration. Third, the metrics should be amenable to management, so that water managers can develop environmental flow standards using these same hydrologic metrics and evaluate the effect of other water uses in the river on these metrics. Hundreds of flow metrics have been published (Richter *et al.*, 1996; Olden & Poff, 2003; Mathews & Richter, 2007) and are potential candidates for inclusion in a regional river classification. In selecting the appropriate variables, we recommend using the method developed by Olden and Poff (2003) contained in the Hydrologic Assessment Tool software of the Hydroecological Integrity Assessment process (Henriksen *et al.*, 2006; Kennen et al., 2007). The software performs a redundancy analysis to determine which variables are the most informative components of the flow regime. Users have flexibility in selecting metrics from suites of inter-correlated variables to choose those that best satisfy the three primary criteria above. In addition, the "environmental flow components" (EFCs) recently added to the Indicators of Hydrologic Alteration software (Mathews & Richter, 2007) are well suited for ELOHA applications due to their strong link between environmental flow assessment and implementation, their ecological relevance, and their intuitive appeal; however, their information overlap with other metrics has yet to be assessed.

Geomorphic Sub-classification At the broad, regional scale of ELOHA, it will be useful to account for some of the dominant environmental factors that can provide a "context" for interpreting ecological responses to flow alteration and thus for guiding development of flow management rules. Geomorphology is of prime interest in this regard (but other factors might be as well; see discussion in next section).

Geomorphic sub-classification of stream or river segments can provide a useful integration of catchment and local geomorphic characteristics such as geology, channel confinement, and channel slope (Seelbach *et al.*, 1997; Higgins *et al.*, 2005). The geomorphic setting of a river segment will strongly influence how the flow regime gets "translated" into the hydraulic habitats experienced by, and available to, the riverine biota. For example, whether a given level of flow will create a bed-moving disturbance or an overbank flow is determined by local geomorphic characteristics such as channel geometry, floodplain height, and streambed composition. In other words, the same level of flow in one geomorphic setting may not translate into an important ecological event, whereas in a second setting it may (Poff *et al*., 2006a). Therefore, differentiating rivers on the basis of geomorphic setting (e.g., constrained channels vs. alluvial channels; sand-bedded vs. cobble-bedded reaches) will contribute to development of flow alteration-ecological response relationships that reflect the direct and indirect influences of hydrologic alteration on both ecological processes and ecosystem structure and function (Snelder & Biggs, 2002; Jacobson & Galat, 2006).

3. Computing Flow Alteration

ELOHA is grounded in the premise that increasing degrees of flow alteration from baseline condition are associated with increasing ecological change. The degree by which each hydrologic variable differs between the baseline and developed condition is calculated for each analysis node using available software (e.g., Henriksen *et al.*, 2006; Mathews & Richter, 2007). This analysis produces a set of hydrologic alteration values expressed as percent deviation from baseline condition for each analysis node, for each of the hydrologic metrics used to define that river type. These values are then used, along with any additional hydrologic variables of management interest, to develop the flow alteration-ecological response relationships that form a basis for developing environmental flow standards.

The ELOHA process calls for modeling hydrographs at ungauged locations, for both baseline and current conditions. Promising approaches (i.e., that are technically feasible and cost-effective) include watershed rainfall-runoff models that use climate and landscape data and account for human alterations. For example, the water evaluation and planning system (WEAP; http://weap21.org) is a GIS-based software platform that uses a rainfall-runoff model to generate unimpaired hydrographs. By incorporating operational rules for water infrastructure, it can also generate current condition hydrographs throughout a stream network, allowing questions of environmental flows to be addressed (Vogel et al., 2007; Yates et al., in press). Another approach, by Kennen et al. (2008), couples runoff modeling for pervious and impervious areas with estimates of annual water extraction, discharges, and reservoir storage. This model was used to generate daily hydrographs (current conditions) at ungauged locations throughout New Jersey. It is useful for estimating unimpaired conditions at ungauged locations, degree of hydrologic alteration, and can be adapted to include hydrologic forecasting. Other watershed hydrology models are used to generate and compare unimpaired and human-altered streamflow (e.g., PRMS, HSPF, HEC-HMS, SHE, and so on); but many such models are parameter-intensive and can be relatively costly to apply. For a comprehensive description and review of these and other hydrologic models that are applicable to watershed management, refer to Singh (1995).

4. Formulating Flow Alteration-Ecological Response Relationships for Environmental Flows

A key element in the ELOHA framework is defining relationships between altered flow and ecological characteristics that can be empirically tested with existing and newly collected field data. These relationships are hypothesized to vary among the major river types, as ecological responses to the same kind of flow alteration are expected to depend on the natural (historic) flow regime in a given geomorphic context.

Ideally, the relationships between ecological variables and degrees of flow alteration would be expressed in a fully quantitative manner (i.e., % ecological change in terms of % flow alteration as measured at multiple sites along a flow alteration gradient – e.g., Arthington *et al*., 2006). However, ecological changes can also be formalized, and empirically tested, when they are expressed as categorical responses (e.g., low, medium, high) or even trajectory of change (+/-). Such categorical or trajectory relationships can often be robustly defended and provide valuable information in guiding management decisions in many cases (e.g. Arthington *et al*., 2003; King *et al*. 2003, King & Brown, 2006; Shafroth *et al.*, this volume).

Developing Flow Alteration-Ecological Response Hypotheses. In this section, we articulate the principles behind developing testable relationships between ecological variables and flow regime alteration that can serve as a starting point for empirically-based flow management at a regional scale. We also point out some key uncertainties in developing such relationships, and we pose these as challenges for near-future environmental flows research.

Riverine scientists possess a very solid, *general* knowledge of how ecological processes and ecosystem structure and function depend on hydrologic variation. The large literature in hydro-ecology is comprised of both comparative and experimental studies that relate ecological processes or aspects of ecosystem structure and function to the quantity of some hydrologic variable(s) (see examples below). However, very few studies have been published where ecological metrics have been quantified in response to various degrees of flow alteration *per se*, because this requires that hydrologic variables be expressed in terms of deviation from some baseline condition for each sampled location, and this has rarely been done (but see Freeman & Marcinek, 2006; Poff & Zimmerman, this volume). Therefore, empirical models that directly predict ecological responses to various types and degrees of flow alteration (the goal of environmental flows science) are not readily available. The development of such models is an important component of the ELOHA framework, and this can be accomplished by posing testable hypotheses based on the many published studies that document the response of ecological processes and patterns to a range of flow conditions, both natural and altered.

A guiding principle for such model development from the existing hydro-ecological literature is that ecological responses to particular components of the flow regime can be interpreted most robustly when there is some *mechanistic* or *process-based* relationship between the ecological response and the particular flow regime component. Numerous examples exist for many combinations of ecological responses and flow components (see Poff et al., 1997; Bunn & Arthington, 2002; Nilsson & Svedmark, 2002; Poff & Zimmerman, this volume, for reviews). For instance, with increasing frequency of high flow disturbances, macroinvertebrate communities shift toward species adapted to high mortality rates, such as those having short life cycles and high mobility (Richards*, et al.*, 1997, Townsend, Scarsbrook & Dolédec, 1997). More frequent flow fluctuations or increased stream flashiness (such as induced by operations of hydropower dams or urbanization) favor fish species with more generalized vs. specialized foraging strategies (Poff & Allan, 1995) or that are habitat generalists (Bain, Finn & Booke, 1988; Pusey, Kennard & Arthington, 2000) or that are more tolerant of stressful inter-flood low flow periods (Roy *et al.*, 2005). Prolonged (and unnaturally timed) low flows can dewater floodplain vegetation and cause more drought-tolerant species to replace riparian species (Leenhouts, Stromberg & Scott 2006) or reduce fast-flow specialist fish species and encourage habitat generalists (Freeman & Marcinek, 2006). Truncation of natural flood peaks can prevent recruitment of indigenous riparian vegetation and allow non-native trees to become established and proliferate (Stromberg *et al.*, 2007) and can facilitate the proliferation of non-native, flood-intolerant fish species (Meffe 1984). The natural timing of flood peaks can prevent the establishment success of non-native fish (Fausch *et al.*, 2001), whereas the loss of such seasonal flooding can promote invasive fish species success (Marchetti & Moyle, 2001) and even modify river food webs (Wootton, Parker & Power, 1996). The magnitude of flood peaks can determine the degree of scouring mortality of fish eggs in streambed gravel (Montgomery *et al.*, 1999), and altering the duration of flooding can modify geomorphic processes such as lateral channel migration (Richter & Richter, 2000). In terms of ecosystem processes, magnitudes of transport of nutrients and suspended organic matter are dictated by frequency and duration components of the hydrograph (Doyle *et al*., 2005). In summary, these clear relationships (and many others) reflect strong linkages between flow and ecological processes in both unmodified and regulated rivers of different types. This information provides a scientifically-sound and empirically robust foundation for flow-based management of streams and rivers at regional scales.

The exploration of relationships between flow alteration and ecological changes begins by posing a series of plausible hypotheses that are based on expert knowledge and understanding of the hydro-ecological literature. In our experience scientists can readily formulate hypotheses that express testable relationships between flow alteration and ecological changes once they are asked to focus on a limited set of hydrologic variables, such as those resulting from Step 3 above. Initial hypotheses describing flow alteration – ecological response relationships can usually be generated fairly readily by scientists working together in a well-facilitated, collaborative setting (see Arthington *et al*., 2004 and Cottingham *et al*., 2002 for comments on expert panel approaches). Indeed, in a workshop among many of the authors of this paper, we quickly generated a number of process-based hypotheses describing expected trajectories of ecological change associated with specific types of flow alteration based on our collective understanding of the literature (Table 1). Similar and more specific hypotheses can reasonably be developed for particular regions by scientists familiar with the ecology and hydrology of a particular region. Assembling experts to develop flow alteration-ecological response relationships will also assist scientists in identifying available ecological datasets and in designing monitoring programs or research projects for validating and refining the relationships.

Compiling Ecological Data to Test Flow-Ecology Hypotheses. A great diversity of approaches exists for describing and measuring ecological responses to flow alteration. Ecological indicators (Table 2) may be categorized in a variety of ways: taxonomic identity, level of biological organization (e.g., population or community), structural contribution (e.g., abundance of individuals or number of species), functional contribution in the system (e.g., trophic level) or traits that reflect adaptation to a dynamic environment (e.g., life-history characteristics or morphological features), and rate of response to temporal change (e.g., how quickly species and communities respond to environmental change or whether they reflect a trajectory or terminus of change). Additionally, ecological processes and biota may respond to flow alteration either directly (e.g., as a reproductive cue) or indirectly through a water-quality or habitat-mediated response (see Bunn & Arthington, 2002 for guiding principles). Associated with these multiple possible response variables is the fact that their response times to flow alteration can vary significantly. For example, mature riparian forests may require decades to respond to a flow alteration (Nilsson & Svedmark, 2002), whereas riparian seedlings and macroinvertebrate communities may do so on an annual cycle. Thus, selecting an appropriate suite of ecological indicators should be guided by consideration of the different timeframes within which specific ecological responses occur relative to particular kinds of flow alteration, as well as on the ability to monitor these various responses over time.

Ideal ecological (including habitat) response variables are 1) sensitive to existing or proposed flow alterations, 2) amenable to validation with monitoring data and 3) valued by society (e.g., a decrease in fish abundance could substantially affect important protein sources for local communities). While we advocate the use of process-based ecological response variables, some composite ecological indices may be useful as well, since they correlate with human-induced changes in streamflow. Examples include the indices of biotic integrity (IBI) for fish (e.g., Fausch, Karr & Yant, 1984; Kennard *et al.* 2006) or benthic invertebrates (e.g., DeGasperi *et al.*, in press), and the Lotic-invertebrate Index for Flow Evaluation (LIFE) scores (e.g., Monk *et al.*, 2007). However, it may be more useful to disaggregate these indices into their component metrics, some of which may represent a mechanistic relationship to flow or habitat. As indicated above, many studies have demonstrated that ecological responses to flow variation and alteration can be inferred when viewed through the prism of the biological attributes of species (e.g., resource and habitat utilization traits or life history traits), and species trait databases are now being compiled regionally to globally for macroinvertebrates (e.g., Usseglio-Polatera et al., 2000; Poff et al., 2006b) and fish (Winemiller & Rose, 1992; Welcomme, Winemiller & Cowx, 2006).

In many cases, developing relationships that link flow alteration to habitat response can provide valuable information in developing regional environmental flow criteria. In particular, where biological data and scientific resources are scarce (e.g., in many developing countries), habitat assessments may provide a critical scientific basis for environmental flows. Approaches to linking flow regime alteration to habitat change are relatively well developed (Bovee *et al.*, 1998; Bowen, 2003; Pasternack, Wang & Merz, 2004; Crowder & Diplas, 2006; Jacobson & Galat, 2006), and they allow some inference about many ecological responses, albeit with some uncertainty (Tharme, 2003; Gippel, 2005). Flow-habitat linkages and their ecological consequences provide a core component of several existing environmental flow methodologies, e.g. DRIFT (Downstream Response to Imposed Flow Transformation: Arthington *et al*., 2003; King *et al*., 2003).

In general, developing characterizations of hydraulic habitat conditions that can be applied at the regional scale depends substantially on a segment-scale geomorphic sub-classification that resolves river reaches with similar channel morphology (as described in Step 2 above). Such geomorphic subtypes would be expected to have similar hydraulic responses to altered flow regimes. Low-intensity hydraulic habitat assessment methods may be applicable to generalize hydraulic habitat relations for specific geomorphic subclasses. For example, Lamouroux (1998), Lamouroux, Souchon & Herouin (1995) and Booker & Acreman (2007) have developed generalized models for depth and velocity at the stream reach scale, and Saraevan & Hardy (in press) presented a method for extrapolating reach-specific habitat data to unmeasured reaches throughout a catchment using a process based on hydrologic and geomorphic stratification. Additionally, applications of habitat-based methods like the wetted perimeter approach (Gippel & Stewardson, 1998), PHABSIM (Bovee *et al*, 1998) or MesoHABSIM (Parasiewicz, 2007) could provide habitat information useful in the ELOHA framework.

Flow alteration – ecological response relationships. The functional relationship between an ecological response and a particular kind of flow alteration can take many forms, as noted by Arthington *et al.* (2006). Based on current hydro-ecological understanding, we hypothesize that the form of the relationship will vary depending on the particular ecological response variable(s), the specific flow metric(s) the degree of alteration from the baseline condition, and the river type. These relationships could follow a number of functional forms, from monotonic to unimodal to polynomial. For illustrative purposes, we will consider three likely general types:

no relationship, a linear response and a threshold response (step function). We summarize various combinations of these likely response functions in Fig. 3. Note that these relationships may vary among different kinds of ecological metrics and thus need not be "symmetrical," i.e., the relationship may be linear with respect to negative flow alteration but exhibit a threshold response to positive flow alteration (Fig. 3). Similarly, the slope or direction of linear and threshold responses could be upward or downward, depending on the particular ecological response variable or flow metric selected. The illustrative responses shown in Fig. 3 are expressed as continuous functions; however, they could also be more generally represented as categorical or trajectory responses to hydrologic alteration, as these kinds of functions also represent testable hypotheses.

One important reason for developing a flow regime classification (step 2) is that the form and direction of an ecological response to flow alteration is hypothesized to be similar within river types and vary among river types. For example, Fig. 4 shows five river types developed for 420 streams with unmodified flow regimes in the United States (from Poff, 1996). The ellipses represent the 90% confidence limits for each river type expressed in terms of two of the flow classification variables (baseflow stability and flood predictability) that are ecologically relevant and amenable to management action. The size of each ellipse represents the natural range of variation for the river type in this 2-dimensional space, and based on these natural differences, we would predict different ecological responses to similar types of flow alteration. For example, the stable groundwater type has a higher degree of baseflow constancy (x-axis) than the perennial flashy/runoff type or the intermittent type. Ecological differences exist between these types of streams (see Poff & Allan, 1995). A flow alteration that introduced fluctuations in baseflow (e.g., below a hydropower dam) would be expected to have a much greater ecological effect in the stable groundwater type than in either of the other two types, because they are already highly variable. Conversely, a stabilization of baseflow conditions would likely induce a large ecological response in the intermittent and perennial types, but not in the stable groundwater type where baseflows are already relatively constant. On the y-axis of Fig. 4, the snowmelt type is distinguished by having a very predictable timing of peak flow. A loss of this seasonality would be ecologically important for the snowmelt type, and possibly for the snow/rain type, but less so for the perennial or stable groundwater systems where high pulse predictability is naturally low.

Compiling existing data will support, in many cases, a statistical analysis of the form of the functional responses illustrated in Fig. 3 and a test of the degree to which such responses differ between river types. Exploring these statistical associations will allow identification of critical information gaps and research needs. For example, the ability to detect a threshold vs. linear response for some ecological response variable along a flow alteration gradient may be difficult because ecological data are missing within some critical range of flow alteration or because a small sample size has insufficient statistical power to detect a threshold response. Such initial outcomes can guide strategies for targeting future field data collection at specific points along the flow alteration gradient to resolve key uncertainties (Arthington *et al.*, 2006).

Toward Setting Environmental Flow Standards

Functional relationships between flow alteration and ecological responses provide critical input for the broader societally-driven process of developing river-type-specific, regional flow standards (see Fig. 1). We expect that establishing standards for limiting the degree of each type of flow alteration for different river types will ultimately depend on the ecological goals set for a region's river types, as well as on the "risk" stakeholders and decision-makers are willing to accept to attain those goals. The degree of acceptable risk is likely to reflect the balance between the perceived value of the ecological goals (e.g., maintenance of fisheries may be of particular interest) and the scientific uncertainties in functional relationships between ecological responses and flow alteration. A "benchmarking" approach (see Arthington *et al*., 2006) can be adopted to help establish an ecologically and societally-acceptable level of risk. For example, where there are clear threshold responses (e.g., overbank flows needed to support riparian vegetation or provide fish access to backwater and floodplain habitat), a benchmark of low ecological risk might allow for hydrologic alteration that does not cross the threshold. For a linear response where there is no clear threshold for demarcating low from high risk, a consensus stakeholder process may be needed to determine acceptable risk. One possible process for setting such risk levels is to use expert panels to identify "thresholds of potential concern" (Biggs & Rogers, 2003; Acreman *et al*., 2008), which establish where along the flow alteration gradient there is agreement among stakeholders (including scientists and managers) that further hydrologic change carries with it unacceptably high ecological risk. This approach incorporates scientifically-credible professional judgment and includes multiple ecological indicators, as is commonly employed in performing river-specific environmental flow assessments based on expert judgment as applied in South Africa (Tharme, 2003), Australia (Arthington *et al.*, 2004; Cottingham *et al*., 2002) and in the Americas (Richter *et al.*, 2006).

We note here that the flow alteration – ecological response relationships developed for various river types can be used by water managers to guide development of flow standards for individual rivers or river segments, or for sub-catchments of individual rivers, not just for entire classes of rivers. Indeed, society may have different ecological goals for different sub-catchments or rivers within a class, and the flow-ecology relationships enable river-specific standard setting by associating different flow targets with different ecological targets.

Challenges of interpreting flow-ecology relationships for water management purposes. In interpreting flow alteration – ecological response relationships, there are some major challenges that must be addressed. First, because ecological responses may be expressed in relation to multiple hydrologic drivers, decisions will have to be made about which relationships are the most important or achievable in a particular management context. One possible way to overcome this challenge would be to consider ecological responses in terms of combined, multivariate descriptions of overall flow alteration (e.g., using principal components analysis as in Black *et al.*, 2005), and a multivariate hydrologic metric may capture the complex response of multiple ecological variables. Such multivariate approaches, may allow identification of specific flow metrics (or other environmental variables) that describe ecological response (e.g., Kennard *et al.*, 2007). Often, however, it will be most desirable to consider ecological responses in terms of independent flow variables that can be directly manipulated in a management context.

Where multiple ecological response – flow alteration relationships are generated, some process will be required to prioritize these in a management context. In the face of multiple possible management targets, "paralysis" can be avoided by keeping in mind the motivating objectives of the selection process for a hydrologic variable selection process. Metrics ideally should have been selected to capture a range of natural hydrologic variability, to be ecologically relevant, and to be amenable to management manipulation. Depending on what the societally-acceptable ecological goals are (Fig. 1), we would imagine selecting those relationships that can be mechanistically interpreted, that are known with confidence, that best define the hydrologic character of the river type and that are especially sensitive to human alteration. For example, stable groundwater streams (Fig. 4) are likely to be sensitive to increases in baseflow fluctuations and seasonally pulsed systems (e.g., snowmelt) are likely to be very sensitive to altered timing of pulses. Such class-specific metrics could represent priority management targets, all else being equal. However, we also stress that many metrics would ideally be considered if the management goal is to promote broad ecosystem function. Ideally, a parsimonious suite of flow metrics will emerge that collectively depict the major facets of the flow regime and explains most of the observed variation in ecological response to particular types of flow alteration in each class of rivers.

Second, development of robust flow alteration – ecological response relationships will need to take into account the role that other environmental factors play in shaping ecological patterns in streams and rivers. The ecological integrity of rivers is certainly known to reflect factors other than flow regime, such as water quality and habitat structure (Poff *et al.*, 1997; Baron *et al.*, 2002; Kennen *et al.*, 2008; Konrad, Brasher & May, in press); however, a quantitative understanding of how flow interacts with these other factors is not yet well developed (e.g., Kennard *et al*., 2007; Stewart-Koster *et al*., 2007). We view this as an important research frontier in environmental flows. We have attempted to minimize this consideration by calling for a geomorphic sub-stratification within hydrologic classes to assist the translation of streamflows into appropriate hydraulic habitat contexts. However, some accounting of other environmental factors will be necessary. This could be done either by further stratification (e.g., based on water temperature or water quality; see Olden & Naiman, this volume) or by including additional environmental variables in the flow-ecology models as statistical covariates, which would allow some determination of the independent and interactive effects of flow alteration on ecological processes and metrics.

Learning by Doing: The Scientist's Long-Term Involvement

An environmental flow "standard" is a statement of flow regime characteristics needed to achieve a certain desired ecological outcome. In the ELOHA framework, environmental flow standards are determined by combining the scientific understanding of flow-ecology relationships with a societally-defined goal of environmental health and a particular level of risk of ecosystem degradation. Flow standards may take the form of restrictive management thresholds, such as maximum limits of abstraction, or active management thresholds, such as specific flow releases from reservoirs (Acreman & Dunbar, 2004). Attempts to establish such regional standards are evolving in several political jurisdictions in the United States, including the states of New Jersey, Missouri and Texas. The State of Michigan has proposed a standard on groundwater pumping that protects fisheries resources for each of 11 classes of streams in the state (MGCAC, 2007). In developing the flow-response lines in Fig. 5, fisheries ecologists examined the range of variation in the biological response across the flow alteration (depletion) gradient and effectively smoothed the statistical scatter to create a trend line with cut-points reached by consensus through a stakeholder process (MGCAC, 2007) comparable to benchmarking (see Arthington *et al*., 2006).

We recognize that assessing the ecological effects of modified flows is only one part of a complex socio-economic-environmental process to decide on the use and protection of a region's water resources. The decision to exploit those resources to any particular level is one that will be taken by governments and stakeholders in the context of their perceived priorities for development and sustainability. In essence, a partnership of managers, scientists and those parts of society that will experience the effects of management actions decides on a redistribution of the costs and benefits of water use within the management area (e.g., Naiman, 1992; Poff *et al.*, 2003; King & Brown, 2006; Rogers, 2006). The scientist's role is to support that decision-making process by accurately and usefully communicating the importance of ecosystem goods and services provided by streams, rivers and wetlands and the ecological and societal consequences that will result from different levels of flow modification represented in the flow-ecology relationships.

Scientists can also assist in implementing flow standards once they have been established. Specifically, the regional approach of ELOHA affords the opportunity to quantitatively incorporate environmental flow standards within integrated water resources and river basin management. ELOHA's hydrologic foundation synthesizes all of the controls – both natural and engineered – on streamflow patterns into one usable database. Thus, it is useful not only for establishing flow-ecology relationships, but also for integrating them into the social decision-making process. Scientists and managers can run the hydrologic model to test various stakeholder-developed scenarios for coordinating and optimizing all geographically referenced water uses in a basin, while maintaining environmental flows. The hydrologic model also can incorporate predicted hydrologic impacts of climate change. Because the model accounts for the cumulative effects of all water uses, it can be used to assess the practical limitations to, and opportunities for, implementing environmental flow targets at any analysis node, or for every node simultaneously. It can be used, for example, to prioritize development of restoration projects, optimize water supply or hydropower generation efficiency, or account for cumulative upstream and downstream impacts in permitting decisions. For basins in which water is already over-allocated, it can help target flow restoration options such as dam re-operation, conjunctive management of ground water and surface water, drought management planning, demand management (conservation), and water transactions (e.g., leasing, trading, purchasing, banking).

Finally, scientists must maintain an active role in adaptively managing environmental flows. New information may be required to refine flow alteration – ecological response relationships where few data presently exist, and to extend the relationships in places where climate change and other stressors expand the types and gradients of flow alteration and ecological response. Effective adaptive management means designing, implementing, and interpreting research programs to refine flow-ecology relationships, and ensuring that this new knowledge translates into updated, implemented flow standards (Poff et al., 2003)

Conclusion

The scientific process and recommendations presented in this paper represent our consensus view for greatly enhancing sustainable management of the world's rivers for ecological and societal benefits in a timely manner and over greater spatial scales than are typically attempted. We recognize that the strength of relationships between flow alteration and ecological response is likely to be subject to various interpretations in many instances. Many relationships are likely to be supported in a trajectory or categorical mode, whereas strong statistical support for incremental or continuous relationships is more difficult to establish. We also recognize that the strength of the relationships necessary to support management or policy action may be a key issue in developing and implementing regional flow guidelines in certain social-political settings.

Despite these acknowledged constraints, the consensus of this group of authors is that the body of scientific knowledge and judgment is strong enough to provide a firm foundation for moving forward. Much remains to be learned, but we know enough to start. One of the key goals of restoration ecology is to "do no harm" and to attempt to achieve ecosystem self-sustainability through management action (Palmer *et al*., 2005). The ecological health of the world's riverine ecosystems is presently so threatened that we posit it is in society's best interest to promote regional environmental flow management to do just that. Further, through future adaptive learning and research the ELOHA framework can provide a foundation for refining efforts to optimize the tradeoffs inherent between resource exploitation and resource conservation (Dudgeon *et al*., 2006).

We have emphasized in this paper that scientific knowledge and theory pertaining to flow alteration-ecological response principles has advanced markedly in recent decades, and the caliber of data and "professional judgment" available to inform relationships between flow alteration and ecological response has vastly improved. Ideally, the ELOHA framework should be used to set initial flow standards that can be updated as more information is collected in an adaptive cycle that continuously engages water managers, scientists and stakeholders to "fine tune" regional environmental flow standards (Fig. 1). The process of setting standards during this first iteration should include recognition of knowledge gaps and the need to quantify ecological responses in key areas and in relation to known risk factors. Subsequent iterations will then be informed by more quantified information as needed to satisfy managers and stakeholders. Importantly, we expect that first-iteration applications of the ELOHA framework will greatly help to inform decision-makers and stakeholders about the ecological consequences of flow alteration, and will generate support for the additional data collection needed to further refine the hydrologic foundation, the flow alteration-ecological response relationships and regional environmental flow standards.

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Table 1. Examples of hypotheses to describe expected ecological responses to flow alteration, which were formulated by the authors of this paper during a 2006 workshop. Scientists applying ELOHA should formulate similar hypotheses for their region of interest as a first step in developing flow alteration-ecological response relationships. Flow categories based on "environmental flow components" from Mathews and Richter (2007).

Extreme low flow

Hyp: Depletion of extreme low flows in perennial streams and subsequent drying will lead to rapid loss of diversity and biomass in invertebrates and fish due to declines in wetted riffle habitat, lowered residual pool area/depth when riffles stop flowing, loss of connectivity between viable habitat patches, and poor water quality.

Hyp: Increased dry spell duration in dryland or intermittent rivers will lead to reduced diversity and biomass of invertebrates and fish due to reduction in permanent, suitable aquatic habitat.

Hyp: Increased duration of extreme low flows will result in canopy die-back in arid to semi-arid landscapes.

Low Flow

Hyp: Depletion of low flows will lead to progressive reduction in total secondary production as habitat area becomes marginal in quality or is lost.

Hyp: Augmentation of low flows may lead to an initial increase in total primary and secondary production but this would decline with drowning of productive riffles and/or increased turbidity and decreased light penetration.

Hyp: Augmentation of low flows will cause a decline in richness and abundance of non-fluvatile species with preferences for slow-flowing, shallow-water habitats, whereas fluvatile or obligate rheophilic species would shift in distribution or decline in richness and abundance if low flows were depleted.

Hyp: Augmentation of low flows will result in increased establishment and persistence of aquatic and riparian vegetation with concomitant shifts in species distributions towards increased dominance by fewer species.

Small floods / high flow pulses

Suggest adding an obvious geomorphological response e.g. sediment aggradation with a loss in high flow events competent to transport finer sediments

Hyp: Lessened frequency of substrate-disturbing flow events leads to shift to long-lived, large-bodied invertebrate species in non-flashy streams.

Hyp: Lessened frequency of substrate-disturbing flow events leads to reduced benthic invertebrate species richness as fine sediments accumulate, blocking substratum interstitial spaces.

Hyp: Increased frequency of substrate-disturbing events leads to a shift toward "weedy" invertebrate species and loss of species with poor re-colonization ability.

Hyp: Increased flood frequency (in channels) will reduce abundance of young-of-the-year fish, but decline in flood frequency would favor flood-intolerant species.

Hyp: A decrease in inter-annual variation in flood frequency (i.e., stabilized flows) will lead to a decline in overall fish species richness and riparian vegetation species richness, as habitat diversity is reduced.

Hyp: Changes in small flood frequency will lead to changes in channel geometry (dependent upon boundary materials)

Large floods

Hyp: Lessened frequency or extent of floodplain inundation will lead to reduced invertebrate and fish production or biomass due to loss of flooded habitat and food resources supporting growth and recruitment.

Hyp: Increases in floodplain inundation frequency will enhance productivity in riparian vegetation species through increased microbial activity and nutrient availability, up to a point of water-logging, after which productivity would decline due to anaerobic soil conditions.

Hyp: Increasing frequency of floodplain inundation will lead to increases in the proportion of riparian plant species.

Table 2. Considerations in selecting ecological indicators useful in developing flow alteration-ecological response relationships.

Mode of response

Direct response to flow
Indirect response to flow, e.g., habitat-mediated

Rate of response

Fast vs. slow
- Fast: appropriate for small, rapidly-reproducing, or highly mobile organisms
- Slow: long life span

Terminal vs. trajectory
- Terminal: reflect “recovery” or “equilibrium” (mostly fast indicators)
- Trajectory: establishment of tree seedlings, return of long-lived adult fish to potential spawning habitat

Taxonomic groupings

Aquatic vegetation
Riparian vegetation
Macroinvertebrates
Amphibians
Fishes
Terrestrial species (arthropods, birds, water-dependent mammals, etc.)
Composite measures, such as species diversity, Index of Biotic Integrity

Functional attributes

Production
Trophic guilds
Morphological, behavioral, life history adaptations (e.g., short-lived vs. long-lived, reproductive guilds)
Habitat requirements and guilds
Functional diversity and complementarity

Biological level of response (process)

Genetic
Individual (energy budget, growth rates, behavior, traits)
Population (biomass, recruitment success, mortality rate, abundance, age-class distribution)
Community (composition; dominance; indicator species; species richness, assemblage structure)
Ecosystem function (production, trophic complexity)

Habitat responses linked to biological changes

Changes in physical (hydraulic) habitat (width-depth ratio, wetted perimeter, pool volume, bed substrate)

Changes in flow-mediated water quality (sediment transport, dissolved oxygen, temperature)
Changes in in-stream cover (e.g. bank undercuts, root masses, woody debris, fallen timber, overhanging vegetation)

Social value

Fisheries production, clean water and other ecosystem services or economic values
Endangered species
Availability of culturally-valued plants and animals or habitats
Recreational opportunities (e.g. rafting, swimming, scenic amenity)
Indigenous cultural values

Fig. **1.** The ELOHA framework includes three parallel processes – hydrologic (blue), ecologic (green), and social (orange) -- linked together by flow alteration-ecological response relationships. This paper describes four steps in the scientific process in detail, and outlines the scientist's role in the social process.

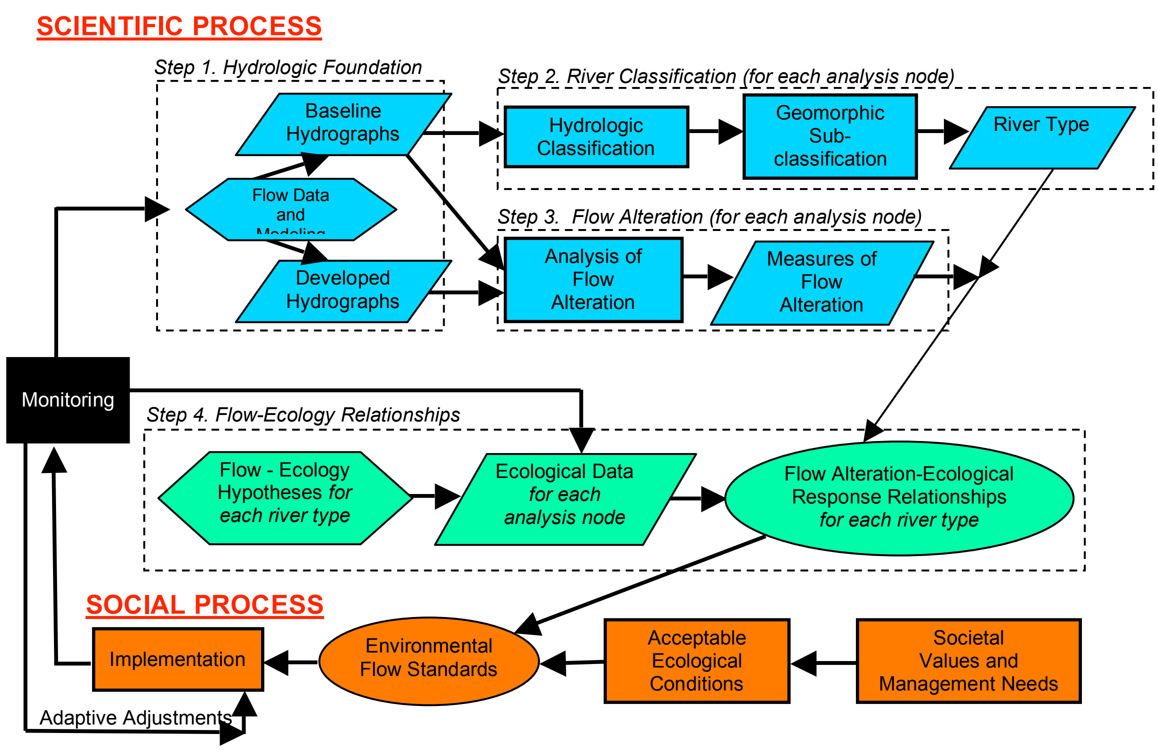

Fig. 2. Steps for developing the hydrologic foundation (ELOHA step 1 inside dashed box), showing how the resulting hydrographs are used to classify river types (ELOHA step 2) and calculate flow alteration (ELOHA step 3) at each analysis node.

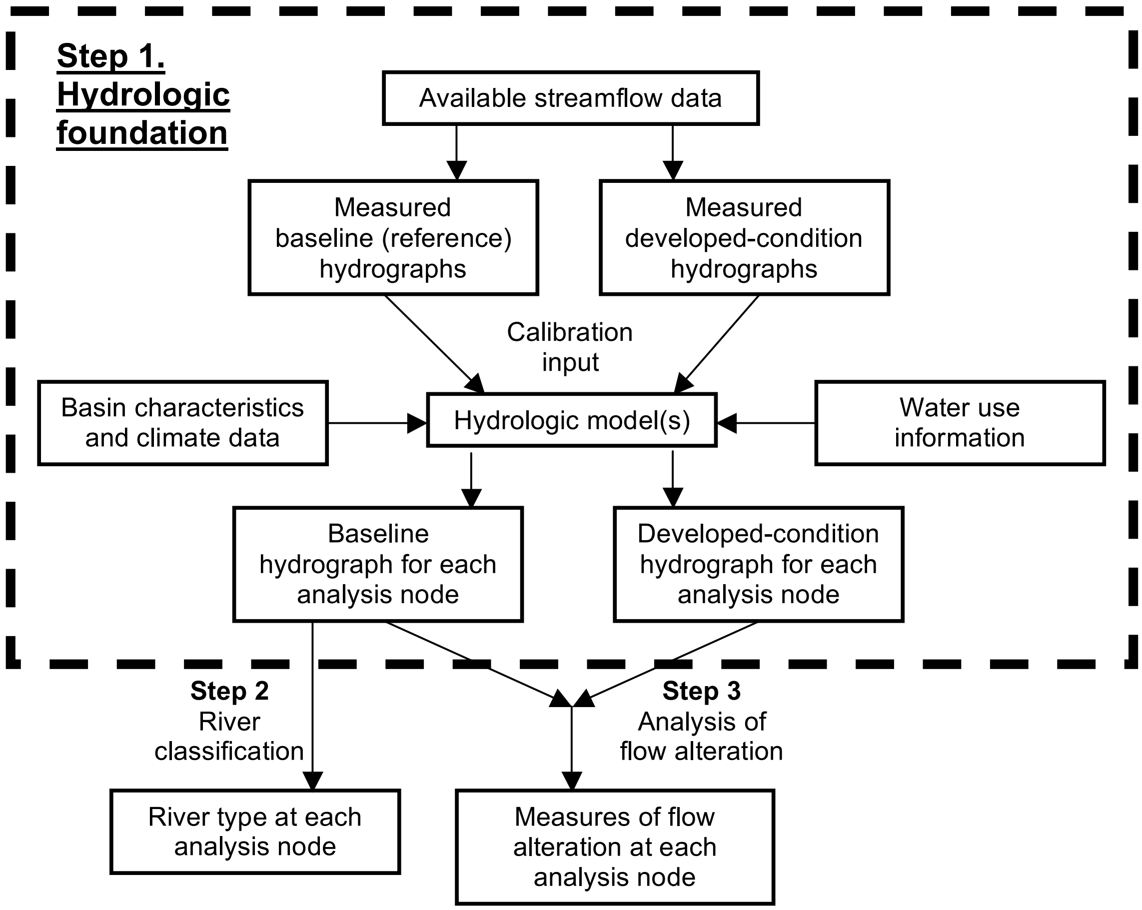

Fig. 3. Six possible forms of functional relationships between a particular combination of an ecological response and a flow alteration variable. For each example, the box in the center of the graph represents the natural range of variation (at "reference" sites) for the flow variable (X-axis) and the ecological variable (Y-axis). Ecological response to both negative (to the left of 0 on the X-axis) and positive flow alteration (to the right of 0) can be any combination of none (flat line), linear (straight sloped line) or threshold (jointed line). Similarly, the slopes or direction of the step function can be down or up, depending on the particular ecological response variable chosen or flow metric selected. See text for further explanation.

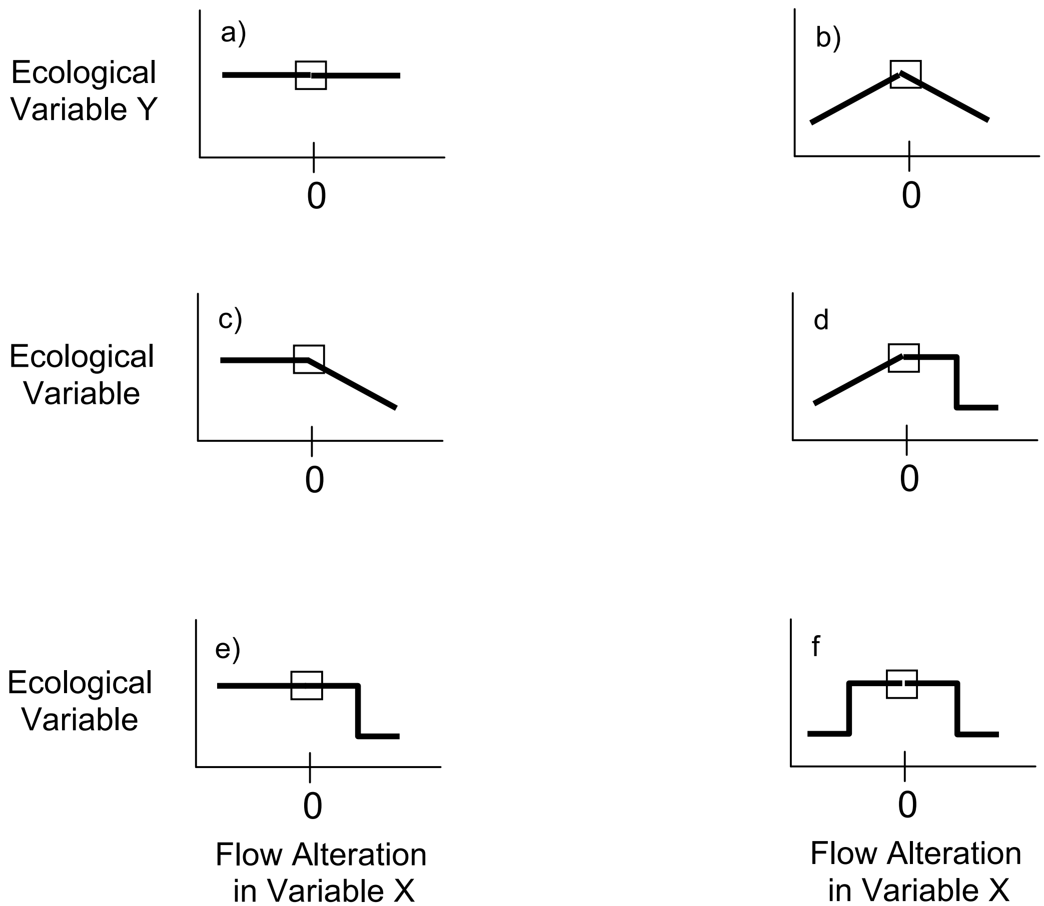

Fig. 4. Plot of five river types in US (modified from Poff, 1996 and Olden & Poff, 2003). River types (based on 420 stream gauges) are defined in terms of 11 flow metrics but plotted here in two-dimensional space defined by two of the classification flow metrics (flood predictability and baseflow index). Ellipses reflect 90% confidence interval for river types and show natural range of variability of the two flow metrics for each river type.

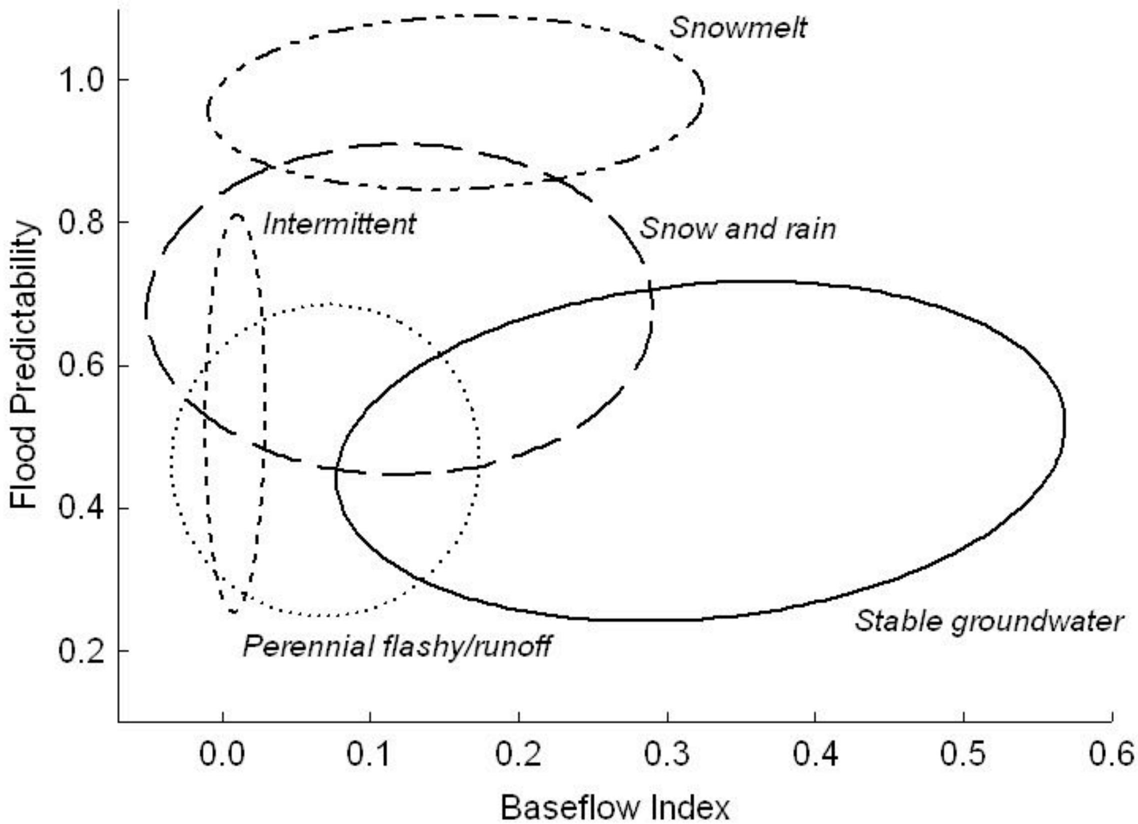

Fig. 5. Progression from flow alteration-ecological response relationships to environmental flow standards (modified from MGCAC, 2007). Using existing fish population data across a gradient of hydrologic alteration, scientists developed two flow-ecology relationships between populations of "thriving" and "characteristic" fish species versus proportion of "index" flow (median August discharage divided by mean annual discharge) flow reduction in 11 stream types in Michigan, USA. A diverse stakeholder committee then proposed a ten percent decline in the thriving fish population index as an acceptable resource impact, and a ten percent decline in the characteristic fish population index as an adverse impact. The corresponding flow alteration (X-axis) would trigger environmental flow management actions associated with each of these ecological conditions. The "ten-percent rule" applies to all of the 11 stream types, but the shapes of the curves – and therefore the allowable degree of hydrologic alteration -- vary with stream type.