REPORT

Yampa-White Basin Roundtable Watershed Flow Evaluation Tool Study

Prepared for:

The Nature Conservancy

Funded by:

Colorado Water
Conservation Board
Water Supply Reserve
Account Grant

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Acronyms

AF	acre-feet
BOR	Bureau of Reclamation
CDSS	Colorado Decision Support System
cfs	cubic feet per second
CWCB	Colorado Water Conservation Board
USDOI	U.S. Department of Interior
ELOHA	Ecological Limits of Hydrologic Alteration
GIS	geographic information system
GVC	geomorphic valley classification
IBCC	Interbasin Compact Committee
IHA	Indicators of Hydrologic Alteration
PBO	Programmatic Biological Opinion
PCI	Potential for Conflict Index
StateMod	State of Colorado Stream Simulation Model
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
WFET	Watershed Flow Evaluation Tool
WSRA	Water Supply Reserve Account

Section 1

Introduction

1.1 Background

The last decade brought many changes to the State of Colorado's water supply outlook. During the past two decades, the state has experienced significant population growth, and Colorado's population is expected to nearly double within the next 40 years. Colorado needs to provide an adequate water supply for its citizens and the natural environment, yet Colorado is transitioning from an era where some water remains to be developed to an era in which we need to manage a more developed resource and make tough decisions about re-allocating water resources among priorities. Meeting the state's municipal, industrial, agricultural, environmental, and recreational water needs will require a mix of local water projects and processes, conservation, reuse, agricultural transfers, and the development of new water supplies, all of which should be pursued concurrently. Ultimately, the future of Colorado—both its vibrancy and its beauty—is dependent on how our water resources are sustained, used, and developed (Colorado Water Conservation Board [CWCB] 2011).

In 2005, the legislature reaffirmed the need to prepare for a future in which water resources are increasing limited by passing the Colorado Water for the 21st Century Act. This legislation established nine basin roundtables and created a voluntary, collaborative process to help the state address its water challenges. The roundtables were organized to represent Colorado's eight major river basins and a separate basin roundtable for the Denver Metro area (Figure 1-1). In addition to the nine basin roundtables, the Colorado Water for the 21st Century Act established the 27-member Interbasin Compact Committee (IBCC) to facilitate conversations between basins and to address statewide issues. The focus of this study is the Yampa-White Basin.

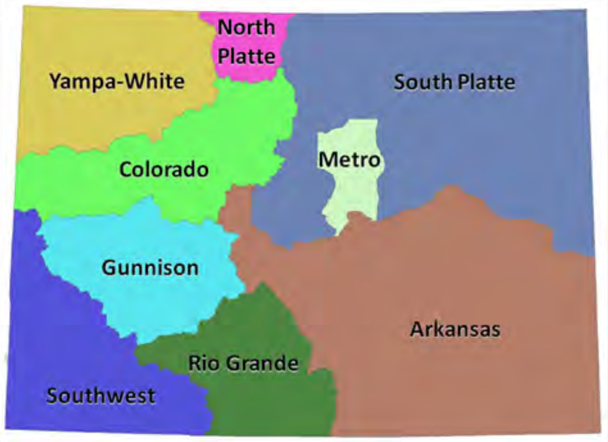

Figure 1-1 Colorado's Nine Basin Roundtables

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
- Proposed projects or methods to meet any identified water needs and achieve water supply sustainability over time

All basins in the state, including the Yampa-White Basin, have followed a similar outline for assessing nonconsumptive needs and identifying projects and methods for meeting those needs (Figure 1-2). The CWCB, who oversees the roundtables, has been working closely with the roundtables as they conduct their assessments and establish projects and methods to meet their nonconsumptive (environmental and recreational) needs. All nine of the basin roundtables have created a list of nonconsumptive attributes for their basin and developed focus area mapping that shows where those attributes occur (CWCB 2011). The Yampa-White Basin's nonconsumptive map and table are included as Table 1-1 and Figure 1-3 at the end of this section. In addition to developing mapping, some basins have quantified water needs for nonconsumptive attributes, and some have studied other aspects of nonconsumptive attributes. A few basins are beginning to describe projects and methods to meet nonconsumptive needs, and it is expected that more basins will be doing so in the coming years. Examples of projects and methods include restoration projects related to improving fisheries, voluntary flow management agreements to address an environmental or recreational need, or CWCB instream flow to protect an environmental need. Based on results from the Colorado Basin's Watershed Flow Evaluation Tool (WFET) efforts, the Yampa-White Basin Roundtable decided they wanted to quantify streamflow needs for their nonconsumptive attributes. Because current methods for streamflow quantification address only a limited number of stream segments and are expensive to implement in multiple locations, the basin roundtable decided to conduct a WFET study in their basin. The WFET offered an approach to assess the flow-related status of nonconsumptive attributes at multiple locations across a watershed. The Yampa-White Basin Roundtable applied for a CWCB Water Supply Reserve Account (WSRA) Grant to apply the WFET throughout the Yampa-White Basin.

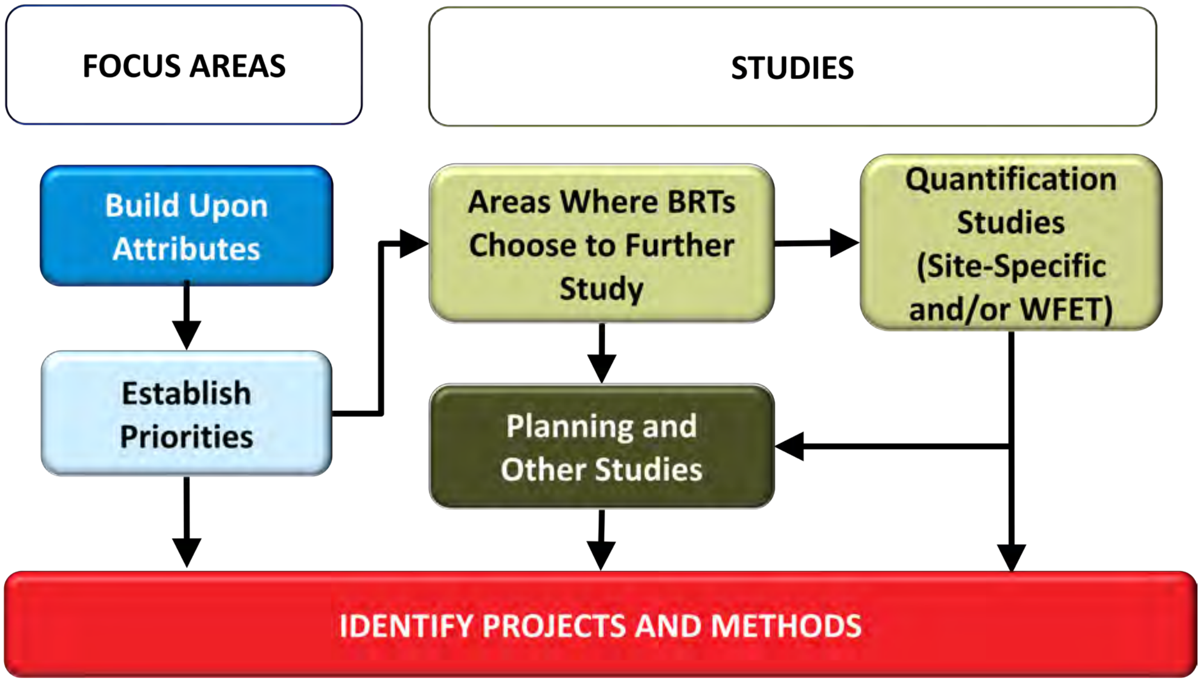

Figure 1-2 State of Colorado Nonconsumptive Needs Assessment Approach

Table 1-1 Yampa/White Basin Nonconsumptive Needs Assessment - Identification of Major Stream and Lake Segments[1]

No.	Stream or Lake Segment (Based Upon Segment Maps)	ATTRIBUTE CATEGORY	1. Federal Threatened & Endangered Fish	2. State Threatened and Endangered Species	3. Important Riparian Habitat	4. Instream Flows and Natural Lake Levels	5. Fishing	6. Boating	7. Waterfowl Hunting	Rationale for Consideration as a Major Segment
					Major Environmental & Recreational Segments					
1	Yampa River - from entrance of Cross Mountain Canyon (East Cross Mountain) to confluence with Green River		a,b,c,d,e	a,b,c,f,e	a,b		c	a	a	Multiple environmental values including critical habitats for endangered fish plus Yampa's most sought after white water and overnight rafting destination including Dinosaur National Monument
2	Yampa River - from Pump Station to confluence of Elkhead Creek			a,c,e,f	a,c		c	a	a	Multiple environmental values plus high use boating and fishing includes TNC's the Carpenter Ranch
3	Green River - from Utah State line (Browns Park Wildlife Refuge) to the Utah State line		a,b,d	a,c,e,f	a,b,c		c	a	a	Multiple environmental and recreational values includes Browns park National Wildlife Refuge and rafting in Dinosaur National Monument
4	Elk River - from headwaters to the County Road 129 bridge at Clark; including the North, Middle and South Fork as well as the mainstem of the Elk			d,f,g	b	a	c	a		Multiple environmental and recreational values including high levels of recreation and significant fisheries use, multiple/critical environmental values
5	White River - from headwaters to Meeker; including the North and South Fork and mainstem of the White			c,d,f	a,b	a	c	a	a	Multiple environmental and recreational values including most extensive, valuable connectivity of Colorado Cutthroat Trout populations in the Yampa/White/Green basin; G1-G3 plant/wetland communities; valuable private and public water fisheries providing significant economic benefits for the upper White basin
6	White River - below Kenney Reservoir dam to Utah State line		b,d,e	a,b,c,f			c	a	a	Multiple environmental and recreational values including critical habitat for endangered fish
					Major Environmental Segments					
7	White River - from Rio Blanco Lake Dam to Kenney Reservoir		b,e	a,b,c				a		Multiple environmental and recreational values including critical habitat for Federal endangered species, multiple state aquatic species of concern
8	**Slater Creek - from headwaters to the Beaver Creek confluence**			**d**	**b**		**c**	**a**		**Valuable connectivity of Colorado Cutthroat Trout populations, with G1-G3 plant communities and multiple recreational opportunities**

Table 1-1 Yampa/White Basin Nonconsumptive Needs Assessment - Identification of Major Stream and Lake Segments[1]

No.	Stream or Lake Segment (Based Upon Segment Maps)	**ATTRIBUTE CATEGORY**	1. Federal Threatened & Endangered Fish	2. State Threatened and Endangered Species	3. Important Riparian Habitat	4. Instream Flows and Natural Lake Levels	5. Fishing	6. Boating	7. Waterfowl Hunting	Rationale for Consideration as a Major Segment
9	**Elkhead Creek - from headwaters to confluence of North Fork of Elkhead Creek**			**a,d**	**b**	**a**		**a**		**Valuable connectivity of Colorado Cutthroat Trout populations, Boreal toad as well as G1-G3 plant communities and recreational opportunities**
10	**South Fork of the Little Snake - from headwaters to confluence of Johnson Creek**			**a,d**		**a**				**Valuable connectivity of Colorado Cutthroat Trout populations**
11	**South and East Fork of the Williams Fork - from headwaters to the confluence of the Forks**			**d,f**	**b**	**a**	**c**			**Valuable connectivity of Colorado Cutthroat Trout populations**
12	**Little Snake River - from Moffat County Road 10 to confluence of the Yampa River**		**c,d**	**b**	**a,b**					**Significant environmental values including occurrences of Colorado Pikeminnow and rare collections of Humpback Chub, populations of Roundtail Chub and valuable riparian plant communities**
13	Yampa River - from Craig (Hwy 394 Bridge) to mouth of Cross Mountain Canyon		d,e	b,e,f	b		c	a	a	Critical habitat for Federal endangered species, multiple state aquatic species of concern
				Major Recreational Segments						
14	Yampa River - from Stagecoach Reservoir "Tailwaters" to northern boundary of Sarvis Creek State Wildlife area			a,c	a	a	c		a	High recreation and fisheries use
15	Fish Creek - from Fish Creek Falls to confluence of the Yampa River				a	a		a		Most significant, highest use kayaking "creek run" in basin
16	Yampa River - from Chuck Lewis Wildlife Area to Pump Station			a,c,e,f	b		c	a,b	a	Highest recreation use along entire Yampa River allowing for multiple recreational opportunities; only RICD in entire Yampa/White/Green Basin
17	Elk River - at Christina State Wildlife Area			c		a	c			Highest public fishery use on Lower Elk River
18	Willow Creek - below Steamboat Lake to confluence with the Elk				a		c	a		Valuable kayaking creek and fisheries use
19	Bear River - from headwaters to USFS boundary			d			c			Cutthroat Trout habitat and significant recreational fishing
20	Stagecoach Reservoir			a			c	a	a	High recreation and fisheries use

Table 1-1 Yampa/White Basin Nonconsumptive Needs Assessment - Identification of Major Stream and Lake Segments[1]

No.	Stream or Lake Segment (Based Upon Segment Maps)	**ATTRIBUTE CATEGORY**	1. Federal Threatened & Endangered Fish	2. State Threatened and Endangered Species	3. Important Riparian Habitat	4. Instream Flows and Natural Lake Levels	5. Fishing	6. Boating	7. Waterfowl Hunting	Rationale for Consideration as a Major Segment
21	Elkhead Reservoir						c	a	a	High recreation and fisheries use
22	Steamboat Lake			d	a		a,b	a	a	High recreation and fisheries use including only Gold Medal Water in basin
23	**Little Snake River - from headwaters of Middle Fork of the Little Snake River and King Solomon Creek to Wyoming border**			**a,c,d**	**b**	**a**	**c**	**a**		**Important fishery including public access and private waters; significant environmental values**
24	**Williams Fork - from South Fork to confluence of the Yampa River**				**a,b**	**a**	**c**			**Important Fishery**
25	**Avery Lake**						**c**		**a**	**Important recreational destination**
26	**Rio Blanco Reservoir**				**b**		**c**		**a**	**Important recreational destination**
27	**Kenny Reservoir**						**c**	**a**	**a**	**Important recreational destination**
28	**Yampa River - Duffy Canyon**		d,e	b,e,f	b		c	a	a	**Important recreational canyon**
29	**Yampa River - Little Yampa Canyon**		d,e	b,e,f	b		c	a	a	**Important recreational canyon**
30	**Yampa River - Juniper Canyon**		d,e	b,e,f	b		c	a	a	**Important recreational canyon**

[1] The CWCB's Statewide Water Supply Initiative Report (http://cwcb.state.co.us/WATER-MANAGEMENT/WATER-SUPPLY-PLANNING/ Pages/SWSI2010.aspx) provides further detail on the data sources used to generate this map.

KEY TO ATTRIBUTE CODES

Attribute 1 - Federal Threatened & Endangered Fish

a. Bonytail Chub
b. Razorback Sucker
c. Humpback Chub
d. Colorado Pikeminnow
e. Federally Listed Critical Habitat

Attribute 2 - State Threatened and Endangered Species

a. Bluehead Sucker
b. Roundtail Chub
c. Flannelmouth Sucker
d. Colorado River Cutthroat Trout
e. River Otter
f. Northern Leopard Frog
g. Boreal Toad

Attribute 3 - Important Riparian Habitat

a. Riparian/Wetland - Dependent Rare Plants
b. Significant Riparian/Wetland Plant Communities
c. Audubon Important Bird Areas

Attribute 4 - Instream Flows and Natural Lake Levels

a. CWCB Instream Flow Water Rights
b. CWCB Natural Lake Level Water Rights

Attribute 5 - Fishing

a. Gold Metal Trout Streams
b. Gold Medal Trout Lakes
c. Significant Fishing Waters (based on local knowledge)

Attribute 6 - Boating

a. Rafting/kayaking/flatwater Reaches
b. Recreational In-Channel Diversion Structures

Attribute 7 - Waterfowl Hunting

a. Waterfowl Hunting

Notes (disclaimer verbiage):

1. Nonconsumptive environmental and/or recreational attributes exist on virtually all stream and lake segments, whether such attributes are identified herein or not. Exclusion of a segment from this chart does not indicate absence of non-consumptive attributes.
2. Attributes associated with the major segments are commonly dependent on conditions in upstream tributary segments. Therefore, the achievement or maintenance of non-consumptive attributes depends upon achieving or maintaining necessary values in upstream segments as well as within the major segment itself.

Important Riparian Habitats were considered based on the following CNHP rankings:

G/S1 Critically imperiled globally/state because of rarity (5 or fewer occurrences in the world/state; or 1,000 or fewer individuals), or because some factor of its biology makes it especially vulnerable to extinction.
G/S2 Imperiled globally/state because of rarity (6 to 20 occurrences, or 1,000 to 3,000 individuals), or because other factors demonstrably make it very vulnerable to extinction throughout its range.
G/S3 Vulnerable through its range or found locally in a restricted range (21 to 100 occurrences, or 3,000 to 10,000 individuals).

1.2 Study Objectives

Following are the study objectives summarized in the WSRA Grant application:

- Develop the WFET in the Yampa-White Basin
- Develop ecological and recreational risk mapping and the associated range of flow for the attributes mapped previously by the Yampa-White Basin Roundtable
- Assess whether water that is being delivered as part of current water rights and Colorado River Compact deliveries in the Yampa-White Basin supports nonconsumptive needs in the basin

The purpose of this report is to summarize the WFET study's approach, results, conclusions, and recommendations. The report is summary in nature and detailed investigations that occurred during the study are provided in the report appendices.

1.3 Report Overview

This report contains the following sections:

- **Section 2 Watershed Flow Evaluation Tool Approach** provides an overview of the WFET, suggested uses for the results in the future, and the methods used in the analysis and validation of results.
- **Section 3 Watershed Flow Evaluation Tool Results** summarizes the results of the analysis and validation.
- **Section 4 Conclusions and Recommendations** presents the conclusions and recommendations of the WFET Study.
- **Section 5 References** includes the previous studies and literature used throughout the study.

1.4 Acknowledgements

The project team would like to acknowledge the hard work of the Yampa-White Basin Roundtable's Nonconsumptive Committee in overseeing the analysis and results of the Yampa-White Basin WFET Project. The committee is led by Geoff Blakeslee and Kent Vertrees.

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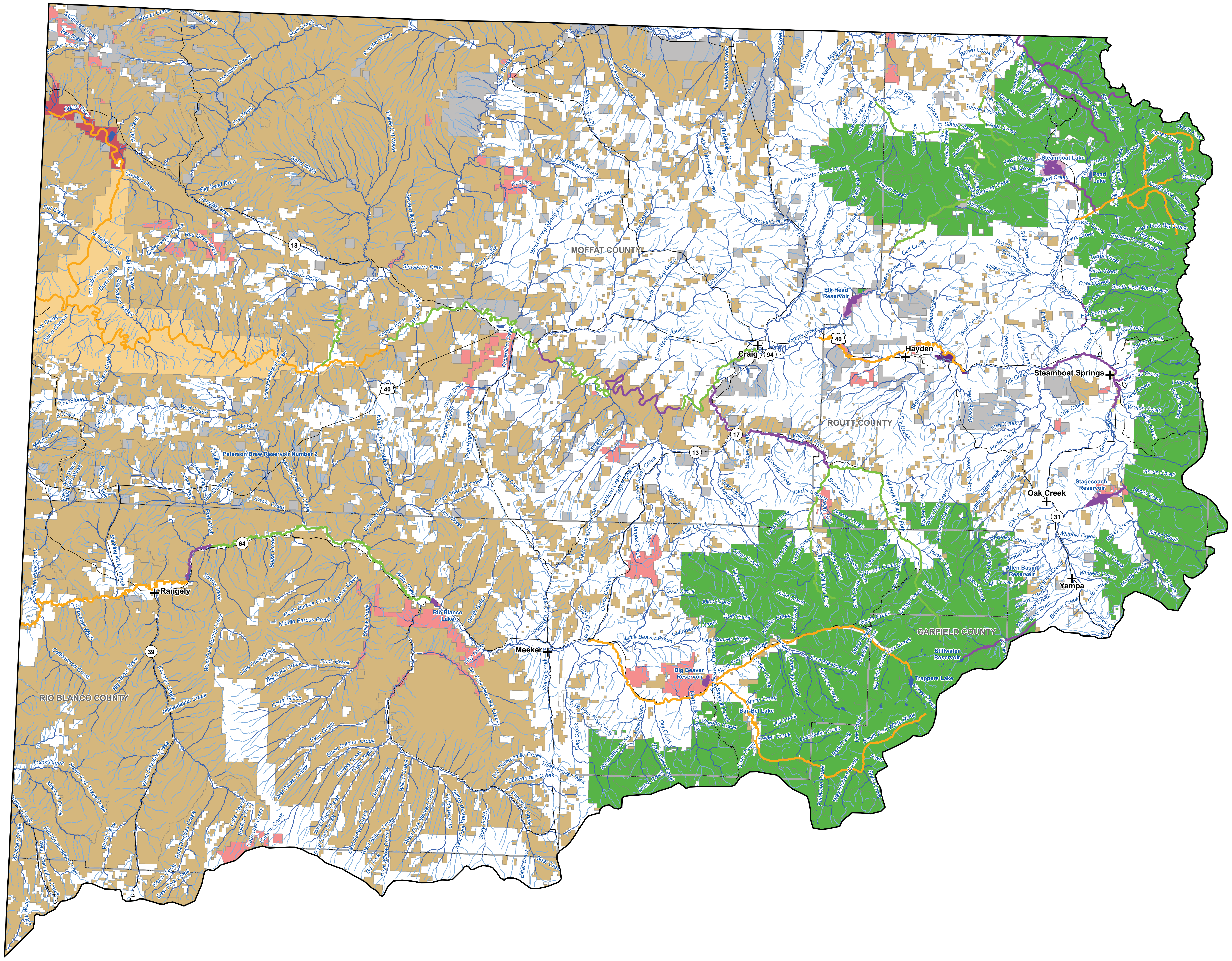

0 4 8 16 24

Miles

1 inch = 4 miles

Document: G:\SWSI\NONCONSUMPTIVE\YAMPA-WHITE\MXD\Figure3-8_YampaWhite_Basin_Env_Rec_MajorSegments_36x48.mxd

Legend

Major Environmental and Recreational Segments

- ~ Environmental Segments
- ~ Environmental and Recreational Segments
- ~ Recreational Segments
- — Roads
- ~ Rivers and Streams
- Lakes and Reservoirs
- ☐ County Boundary

Land Management

- 🟫 BLM
- 🟦 BOR
- 🟥 CDOW
- 🟨 CITY
- 🟩 COUNTY
- 🟥 FWS
- 🟪 LAND TRUST
- 🟧 NPS
- ⬜ PRIVATE
- 🟦 SCHOOL DISTRICT
- ⬜ SLB
- 🟥 STATE
- 🟪 STPARKS
- 🟩 USFS

Figure 1-3
Yampa/White Basin
Nonconsumptive Needs Assessment
Major Environmental and Recreational Segments

Colorado Water Conservation Board
State of Colorado
1876
1937

Refer to Appendix B of the NCNA Mapping Report for a complete list of data sources and Appendix D of the Mapping Report for other basin-specific mapping information.

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Section 2

Watershed Flow Evaluation Tool Approach

2.1 Watershed Flow Evaluation Tool Approach Overview

The Yampa-White Basin Roundtable has expressed interest in quantifying the flows needed to sustain their nonconsumptive attributes. Several long-standing methods exist for quantifying water needs for recreation and the environment, but these methods are: (i) designed for assessing individual river segments, (ii) primarily oriented toward fish (i.e., they did not address other ecosystem needs such as maintaining riparian areas), and (iii) expensive to implement (currently $50,000 to $75,000 for results applicable to tens of kilometers), making it cost-prohibitive to apply them across all streams and rivers in a watershed. As discussed in Section 1, to fill the need for a broadly applicable assessment of flow related to nonconsumptive attributes, the Yampa-White Basin Roundtable has used CWCB's WSRA Grant Funds to complete the WFET[1] study. This study provides a regional framework for understanding flow-related ecological risk for environmental attributes and establishes a baseline for recreational flow needs in the Yampa-White Basin. A regional approach was of interest to the Yampa-White Basin Roundtable because the time and expense of conducting detailed site-specific quantification studies would necessarily limit the studies to just a few locations that would represent at most tens of miles of stream. The Yampa-White Basin has an area of approximately 10,500 square miles and contains about 4,600 miles of named streams (U.S. Geological Survey [USGS] National Hydrography Dataset 2012).

The WFET approach is based on the principle that flow regime is a primary determinant of the composition, structure, and function of aquatic and riparian ecosystems for streams and rivers (Poff et al. 1997). Environmental flows are defined as "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" (Poff et al. 2010 and Appendix A). Environmental flows address specific components of the hydrograph that support specific environmental attributes, including a variable flow regime (Figures 2-1 and 2-2) rather than simply a minimum low flow. Figure 2-1 illustrates several components of a hydrograph that are tied to ecological function. Low flows are needed to maintain aquatic habitat. Seasonal high flows are often needed to flush fine sediment and cue spawning of certain types of fish. Flood flows are needed to sustain riparian ecosystems, scour the channel, and to maintain alluvial water storage (Postel and Richter 2003). The portions of the flow regime related to ecological attributes of the Yampa River at Maybell, Colorado are illustrated in Figure 2-2.

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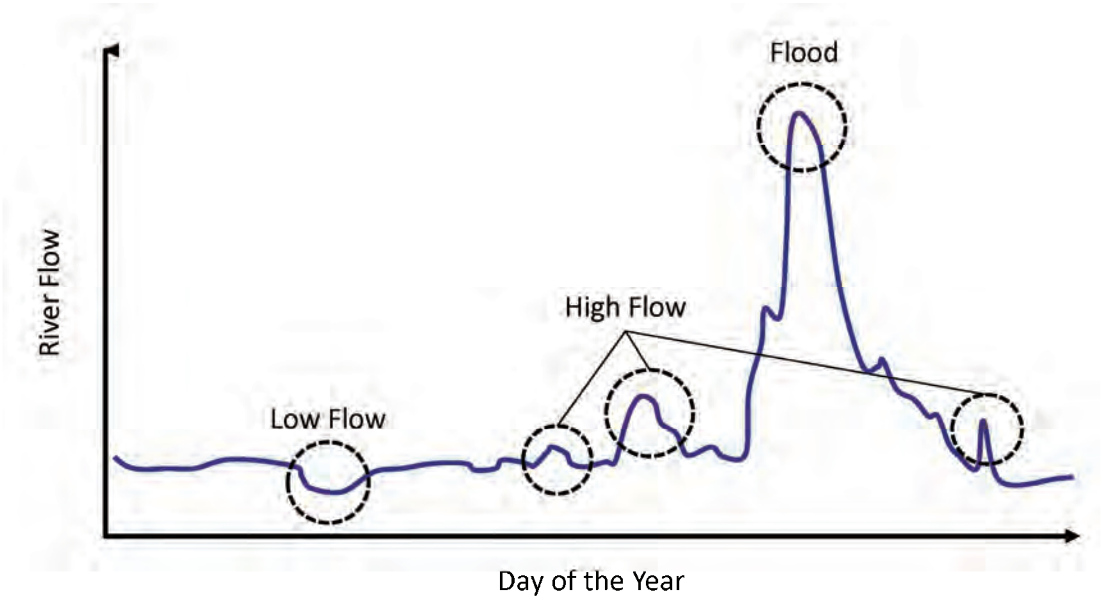
[1] Development of the Watershed Flow Evaluation Tool generally followed the framework presented by Poff NL, Richter BD, Arthington AH, Bunn SE, Naiman RJ, Kendy E, Acreman M, Apse C, Bledsoe BP, Freeman MC, Henriksen J, Jacobson RB, Kennen JG, Merritt DM, O'Keeffe JH, Olden JD, Rogers K, Tharme RE, Warner A. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology* **55**: 147-170.



Modified from Rivers for Life
(Postel and Richter 2003)

Figure 2-1 Example of Environmental Flow Hydrograph.
Low flows are needed to maintain aquatic habitat. Seasonal high flows are often needed to flush fine sediment and cue spawning of certain types of fish. Flood flows are needed to sustain riparian ecosystems, scour the channel, and to maintain alluvial water storage

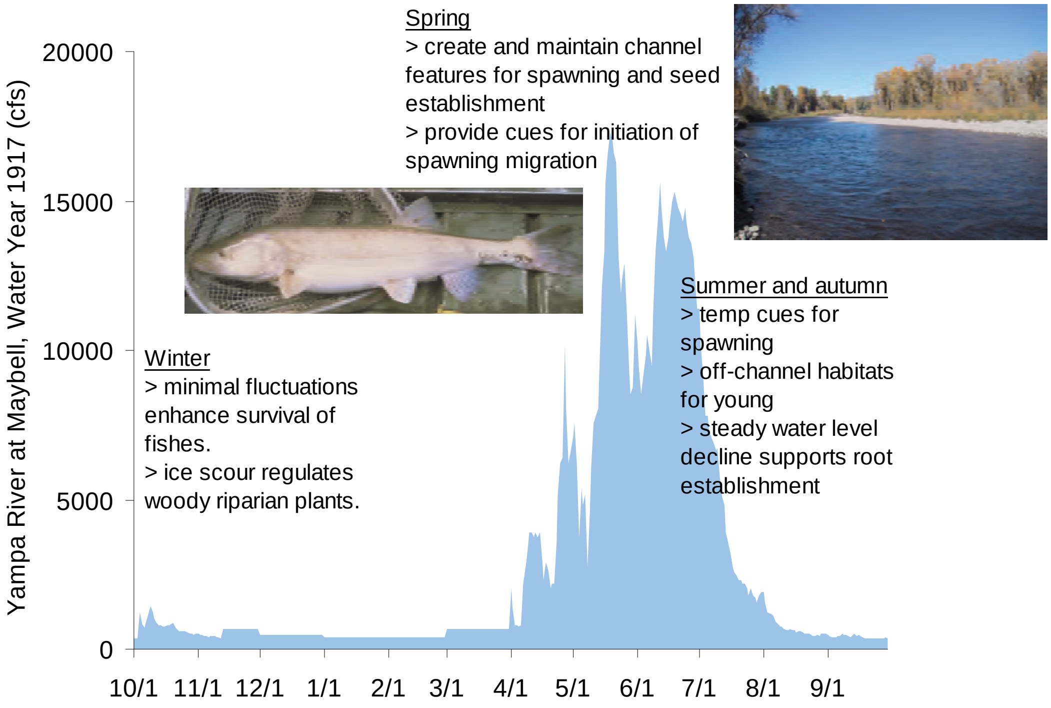


Figure 2-2 Example of Attributes Supported by Environmental Flows in the Yampa River

The WFET is based on the Ecological Limits of Hydrologic Alteration (ELOHA) (Poff et al. 2010, Sanderson et al. 2011) framework for assessing and managing environmental flows across large regions when lack of time and resources precludes detailed (or intensive) field evaluations of all rivers individually. ELOHA uses information from rivers that have been studied and translates this to rivers that have not, without requiring detailed site-specific information for each river (The Nature Conservancy 2011). The scientific basis for ELOHA was published in 2006 by an international group of river scientists (Arthington et al. 2006). Practical guidelines for its application have been developed by consensus of leading international environmental flow experts (Poff et al. 2010).

Table 2-1 describes the steps of the ELOHA Framework (The Nature Conservancy 2011) and how these steps were adapted for the WFET for the Yampa-White Basin. Development of the Yampa-White Basin WFET generally follows the ELOHA framework steps but varies in step 5 as this WFET study is intended for use in water supply planning efforts and not to establish policy in Colorado.

Table 2-1 ELOHA Framework and Application in Yampa-White Basin WFET Study

ELOHA Framework Steps	Colorado Basin WFET Steps
Step 1: Building a hydrologic foundation of daily streamflow hydrographs representing at least two conditions – natural (pre-development) and present-day – for a single time period for every analysis point within the region	Step 1: Hydrologic Foundation. This step is identical to ELOHA's Step 1. The Colorado Decision Support System (CDSS) StateMod model for the Yampa and White Rivers was utilized to develop the hydrologic foundation for the Yampa-White Basin WFET.
Step 2: Classifying river types according to hydrologic and other characteristics	Step 2: Geomorphic Subclassification. This step is similar to ELOHA's step 2. Rivers in the Yampa-White Basin were not classified based on hydrological characteristics as all streams are considered snowmelt driven. A geomorphic subclassification was utilized as part of the Yampa-White Basin WFET. This subclassification was developed to describe the key geomorphic factors that influence riparian systems across large regions.
Step 3: Assessing flow alteration from natural conditions at every analysis point	Step 3: Calculate Flow Metrics. The step is similar to ELOHA's step 3. Natural and current conditions flows were developed for the following flow metrics using The Nature Conservancy's Indicators of Hydrologic Alteration (IHA) software: August mean flow, September mean flow, 90-day maximum flow for wet years, 30-day minimum flow, and maximum average daily flow were calculated as these metrics are utilized in the flow-ecology relationship assessment.
Step 4: Determining flow-ecology relationships that quantify biological responses to different degrees of hydrologic alteration for each river type, based on current biological and related data and models	Step 4: Develop Flow-Ecology and Flow-Recreation Relationships. This step is similar to ELOHA's step 4. For the Yampa-White Basin WFET, flow-ecology relationships were developed for trout, cottonwood, and warm water fish. Flow-ecology relationships are applied only in specific geomorphic settings.
Step 5: Implementing policies to maintain and restore environmental flows through a social process involving stakeholders and water managers informed by the flow-ecology relationships	Step 5: Develop Ecological Risk Mapping. This step in the Yampa-White Basin WFET effort varies from the ELOHA approach. The Yampa-White Basin WFET was developed for use in water planning efforts and has not been utilized to implement policy in Colorado.

The Yampa-White Basin WFET study has also examined the endangered fish biological opinions and related flow recommendations as part of this study. The methods used for this analysis are described in Section 2.7. These methods did not contribute any new information to current knowledge of flow needs of endangered fish; rather the methods simply summarized endangered fish flow recommendations in a manner such that they could be compared to WFET results. The methods used to analyze flows for endangered fish were modeled closely after the Colorado River Basin Water Supply and Demand Study (U.S. Department of Interior [USDOI], Bureau of Reclamation [BOR] 2012).

Recreational flow status was also examined as part of the study effort. The recreational aspects of the Yampa-White Basin WFET study has built upon work conducted by American Whitewater through the United States in developing ranges of flow suitable for whitewater boating. The methods used to examine recreational flow needs in the Yampa-White Basin are described in Section 2.8.

2.2 Applications, Capabilities, and Limitations of the WFET for Ecological Attributes

WFET, as applied in this investigation, is used to assess the risk that stream-based ecological resources may have changed as a result of human uses and the diversion of water, or may change under a scenario in which future streamflows depletions described in the Yampa River Programmatic Biological Opinion (PBO) (U.S. Fish and Wildlife Service [USFWS] 2005) occur. The WFET can help identify watershed areas where the historical or future alteration of streamflow is most likely to have modified ecological resources from conditions that may have historically existed prior to the time that water was first diverted for irrigation, domestic use, and other purposes. The WFET can also be used to examine ecological responses to future streamflow scenarios resulting from new water development projects, a compact call, or climate change.

Flow is considered a "master variable" that is of central importance in maintaining river health (Poff et al. 1997). At the same time, natural influences on ecological resources may include the physical, chemical, geological, and biological properties of the watershed, local climatic conditions, and other related factors such as fire and tree mortality (insect/disease). Anthropogenic activities such as fisheries management, land use practices, physical disturbance, stream channelization, and nonpoint source runoff may also influence ecological resources. The variables that influence ecological resources may be directly or indirectly related to streamflow, or may be unrelated to streamflow. The WFET evaluates the relationship between streamflow and ecology, but does not explicitly consider the other variables, conditions, and interactions not related to streamflow, which can influence the sensitivity of an ecological resource to change.

For many tens of locations throughout a watershed where natural and managed flows have been modeled, the WFET identifies the relative probability that the state of an ecological resource may have changed due to long-term changes in flow, i.e., the WFET evaluates the risk of a change in the river ecosystem resulting from changes in flow. Because of the complex nature of river ecosystems, if the WFET analysis identifies that an ecological resource may be at risk of change as a result of hydrologic alteration, it does not necessarily indicate that an actual change in the ecological resource has occurred, or that any such ecological change that has occurred is specifically attributable to flow alteration.

Using flow metrics to assess the viability of an ecological community necessitates certain assumptions, and the validity of these assumptions can affect the reliability of the results of the WFET. Some of these assumptions are:

- Flow regime is one of the primary determinants of the structure and function of aquatic and riparian ecosystems. This assumption is well-supported by copious peer-reviewed literature spanning well over two decades.

- Modeled streamflows for natural, current, and future conditions are accurate. StateMod was used in the WFET because it is the best hydrologic model available that extends over the entire basin area. Accuracy is expected to be high in some locations and lower in others. Where accuracy is low, additional site-specific measurements of hydrologic conditions may be warranted.

- The 50-year study period for which streamflow estimates have been developed is representative of the long-term climatic conditions to which the ecological resources in the study area are adapted. Several researchers have investigated this assumption, and they have concluded that a 30-year period of record is sufficient to characterize climatic conditions as well as the year-to-year variability inherent in streamflows (Kennard et al. 2009). While the ecological attributes that modeled in the WFET process are important in their own right, there is an assumption that these attributes are also indicators of potential changes in diverse ecological systems, e.g., that cottonwoods also represent other riparian species and that trout also represent other fish.

- Flow-ecology relationships accurately represent the response of the ecological attributes to a change in flow conditions. The flow-ecology relationships are based on current best available science.

Based on the key assumptions outlined above, and upon findings of the WFET pilot studies and comparison with limited site-specific information, the primary capabilities and limitations of the WFET are summarized below.

Capabilities

- The WFET can provide a regional assessment of the risk of ecological change from streamflow alteration, identifying locations with minimal to high risk of change based on flow conditions for specific stream attributes without detailed site-specific information.

- The WFET can identify important seasonal streamflow conditions that may be associated with a risk of ecological change.

- The WFET can be used to target areas that may need further site-specific studies.

- The WFET can be used to identify areas with environmentally healthy flow conditions where nonflow restoration efforts are especially warranted if there are ecological impairments at that location.

- The WFET can help facilitate discussions on a watershed level regarding social preferences and priorities relating to natural resource management and nonconsumptive needs.

- The WFET can be used to assess the vulnerability to ecological change from large-scale water-management scenarios, including major new water development projects, the effects of a Colorado River compact call, benefits or risks associated with a water bank, or future hydrology under climate change scenarios.

- The WFET can be used to identify watersheds with concentrations of "low risk" streams. In these areas, there may be, for example, increased chances of long-term maintenance of environmental goals, because larger connected stream networks are more resilient disturbance.

- The WFET may be used by water providers in the initial planning stages of project development to help determine which project or operation alternative is likely to have the fewest red flags associated with it and/or which may help the environment.
- Although the WFET does not assess or identify any conflicts between recreational and ecological needs, it can potentially be used to explore ways that management scenarios can be crafted to support both recreational and environmental needs.

Limitations

- Because the WFET does not require site-specific ecological data to identify the potential risk of ecological change, it should not serve as the basis for reach-specific flow prescriptions in administrative or judicial processes, absent site-specific data.
- The WFET has been developed to identify the risk of ecological change due to flow alteration, but is insufficient to quantify nonconsumptive water needs on a site-specific basis. Also, the WFET is only one tool in the toolkit for assessing environmental condition as it relates to flow management.
- The WFET will not provide results as detailed or as accurate as a site-specific analysis.
- The WFET does not identify areas where ecological change may be associated with factors other than streamflow, and the WFET does not explicitly evaluate or consider these additional factors that influence ecological and recreational resources, although some of these factors are implicitly considered in the flow-ecology relationships.
- The WFET does not speak to the value of a given change in a resource. For example, it does not address whether or not a change in cottonwood establishment is desirable or not. Rather, the WFET indicates the risk of a change.
- Due to the complexity of determinant factors and ecological response, the WFET does not predict the structure and function of an ecological community under past or future conditions.

2.3 Hydrologic Foundation

The hydrologic foundation for the Yampa-White Basin WFET was developed using the Yampa River Basin Water Resources Planning Model and White River Basin Water Resources Planning Model. These two models are 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. The only modification made to the two models was for the Yampa River Model to represent the releases from Elkhead Reservoir for endangered fish in the lower Yampa River (detailed in Appendix B). For the WFET, the Yampa and White River models were utilized to generate the natural (i.e., human influences removed, referred to as "baseline" in StateMod documentation) and current conditions for flows. For the Yampa River, year 2045 demand datasets available from the CWCB that represent future depletions similar to the Yampa River PBO were used to model future conditions.

StateMod is a water allocation model that simulates the availability of water to individual users and projects based on hydrology, water rights, and operating rules and practices in the Yampa and White River Basins. 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 2-3 at the end of this section shows the distribution of the nodes available to form the hydrologic foundation for this study.

StateMod is capable of simulating both short-term (daily) and long-term (monthly) water allocation conditions. The version of StateMod utilized for the Yampa-White Basin WFET effort was 12.29.30 dated 2/4/2010. The time period that both models covers is water years 1909 – 2005 (October 1908 to September 2005) for monthly simulation and water years 1954 – 2005 (October 1953 to September 2005) for daily simulation (Yampa River Model only). More detailed information regarding StateMod and Yampa and White River Models can be obtained in the CDSS website: http://cdss.state.co.us.

To generate natural flow conditions, the inputs to the Yampa River model were changed to turn off the diversions, instream flow rights, and reservoir operations in the basin. Daily model simulations were performed (Yampa River only). Table 2-2 summarizes the inputs with associated changes. For the White River model, daily operations are not available and the monthly baseflow inputs included with the model files were utilized for this study.

Table 2-2 Summary of Model Inputs with Changes for Simulating Natural 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

2.4 Geomorphic Subclassification

The Upper Colorado River Basin (including the Yampa and White River Basins) contains a diverse mosaic of geomorphic settings and fluvial riparian ecosystems. From the steep, v-shaped, and glacial valleys of the high country to the gentle gradients and expansive floodplains of lowland alluvial valleys, geomorphic setting mediates the relationship between hydrology and riparian ecosystems. Colorado State University has collaborated with the U.S. Forest Service over the last 4 years in the development of a geomorphic valley classification (GVC) for describing the key geomorphic factors that influence riparian systems across large regions. The classification is geographic information systems (GIS) based and delineates different geomorphic valley settings using energy, hillslope coupling, and lateral confinement as the primary diagnostic characteristics. The GVC derives its class descriptions from geomorphic thresholds corresponding to significant transitions in the physical processes and boundary conditions that give rise to distinct floodplain and channel forms, disturbance regimes, and ecological attributes. The GVC was utilized to determine which riparian metric to apply at a given location and to determine if applying the warm water fish metric was appropriate. The GVC classification is only available for the Yampa River Basin. Further information on the GVC methodology is provided in Appendix C.

2.5 Flow Metric Calculations

Certain flow metrics can be considered ecologically important (Olden and Poff 2003). For this study the following flow metrics were determined to be relevant to one or more of the nonconsumptive needs assessment attributes defined in the basin and therefore were calculated at each node where flow data were available:

- Mean annual flow
- Mean August flow
- Mean September flow
- Mean annual peak daily flow
- 30-day low flow (July through November)
- 90-day maximum flow in wet years

IHA software (Richter et al. 1996) was used to calculate these flow metrics for the natural and current conditions datasets outputs from the Yampa and White Basin Models. These flow metrics were selected out of 67 statistical parameters (Richter et al. 1996) to accommodate the calculation of the ecologically relevant flow statistics as presented in remainder of this section.

2.6 Flow-Ecology Relationships and Flow-Ecology Risk Mapping

The flow-ecology relationships were initially developed in the WFET pilot study for the Roaring Fork and Fountain Creek watersheds completed by CWCB in 2009 (CDM Smith et al. 2009; Sanderson et al. 2011). For this study, the flow-ecology relationships from the pilot were reviewed and updated for the following attributes:

- Trout (Appendix D)
- Warm Water Fish (Appendix E)
- Riparian Vegetation (Appendix F)

Based on the hydrologic foundation discussed above and the flow-ecology relationships developed for this study, flow-ecology risk maps were developed. On these maps, the flow-ecology risk was displayed only if the roundtable nonconsumptive attribute map listed trout, warm water fish, or riparian areas in the given segment. This section describes the flow-ecology relationships and the approach for mapping flow-ecology risk. Results of this analysis are presented in Section 3.

2.6.1 Trout Flow-Ecology Relationships

The flow-ecology metric for trout was developed in the WFET pilot study as discussed previously. As part of the Colorado Basin WFET study (Sanderson et al. 2012), the flow-ecology metric for trout was reviewed by comparing the metric with site-specific physical habitat studies. This effort is summarized in Appendix D. The flow-ecology metric for trout is based on a categorical rating of low-flow suitability for trout (cutthroat, brook, brown, and rainbow), from Binns and Eiserman (1979). The flow-ecology relationship is based on summer flows (average for August to September) and is expressed as a percent of natural mean annual flow using the following equation.

$$\frac{(\text{Mean August } Q_{\text{current}} + \text{Mean September } Q_{\text{current}}) \div 2}{\text{Mean Annual } Q_{\text{natural}}} \times 100$$

where:

Q=flow (cubic feet per second [cfs])

This metric was applied in the locations identified in the basin roundtables focus area mapping effort as shown in Figure 1-3 and Table 1-1. Using percentages produced by the above equation, the CDSS nodes were assigned different colors based on the following risk classes for trout:

- <10 percent: Red node color. Low flows are inadequate to support trout (very high flow-ecology risk)
- 10 to 15 percent: Orange node color. Low flows have potential for trout support is sporadic (high flow-ecology risk)
- 16 to 25 percent: Yellow node color. Low flows may severely limit trout stock every few years (moderate flow-ecology risk)
- 26 to 55 percent: Blue node color. Low flows may occasionally limit trout numbers (minimal flow-ecology risk)
- >55 percent: Green node color. Low flows may very seldom limit trout (low flow-ecology risk)

2.6.2 Warm Water Fish Flow-Ecology Relationship

As summarized in Appendix E, the flow-ecology metric is represented by the following equation:

$$\%\text{ maximum native sucker potential biomass}=0.1025\text{ x30-day min flow}^{0.3021}$$

where '30-day minimum flow' is a running mean calculated over the summer-autumn flow period (July 1 to November 30) for each year, then averaged over the study period. In this manner, biomass is estimated for both natural conditions and current flow conditions. Percent reduction in biomass is then calculated as:

$$\%\text{ reduction in potential biomass}=\frac{(\text{natural-current})}{\text{natural}}\times 100$$

The CDSS nodes were assigned different colors based on flow-ecology risk and differentiation among risk levels were derived directly from the flow-ecology relationships for warm water fish as defined above. Risk levels were assigned as follows based on expert recommendations:

- 50 to 100 percent reduction in potential biomass – nodes were assigned a red color (very high flow-ecology risk)
- 25 to 50 percent reduction in potential biomass – nodes were assigned an orange color (high flow-ecology risk)
- 10 to 15 percent reduction in potential biomass – nodes were assigned a yellow color (moderate flow-ecology risk)
- <10 percent reduction in potential biomass – nodes were assigned a green color (low flow-ecology risk)

2.6.3 Riparian Vegetation Flow-Ecology Relationship

The WFET pilot study for the Roaring Fork Watershed developed quantitative relationship between flow alteration and riparian vegetation using many literature sources. The source literature covered a diverse range of vegetation types, including cottonwood, willow, and herbaceous plants. In response to feedback received on the pilot, as well as peer-review comments received during and after an expert workshop, the approach was refined and narrowed as described in detail in Appendix F. This section summarizes that detail. Specific changes and refinements to the methods used in the Roaring Fork WFET pilot include:

- Quantitative flow-ecology relationships were developed for the two riparian types—
  i) cottonwoods on low- and moderate-gradient, meandering (open, or unconfined) rivers,
  ii) cottonwoods in moderate-gradient rivers of confined valleys and high-gradient rivers in unconfined valleys. Despite some evidence of willow dependence on floods (Cooper et al. 2006), we lacked sufficient data to quantify this dependence over a range of flow alteration. For willows, the flow ecology relationship is described only conceptually in Appendix F.
- Flow-ecology relationships are now applied only in the specific elevation ranges and select geomorphic settings where that relationship is expected to exist.
- A new, large data set on cottonwoods (Merritt and Poff 2010) allowed for development of a robust quantitative flow-ecology relationship for cottonwoods in low-gradient, unconfined geomorphic settings. In these settings, flood magnitude alteration is calculated only in 30 percent of years with the highest mean annual flow.

For cottonwood in unconfined geomorphic settings, the attribute was applied for CDSS node locations with a geomorphic setting of moderate-energy unconfined, low-energy floodplain, and glacial trough. In addition, the metric was not applied in locations above 8,700 feet in elevation. Two quantitative flow-ecology relationships exist for cottonwood in unconfined settings—one for adult cottonwood abundance and the other for cottonwood recruitment. The hydrologic metric for adult cottonwood abundance is the change in average 90-day maximum flow in wet years only between current and natural scenarios. "Wet years" are those in the top 30th percentile for mean annual flow in the natural flow time series. Cottonwood abundance is calculated as:

$$\%\ \text{abundance} = 1.038 \times \%\ \text{flow alteration} + 1.005.$$

For cottonwood abundance, the CDSS nodes were assigned different colors based on the following flow-ecology risk classes:

- Flow alteration of 50 to 100 percent was assigned a red node color representing very high flow-ecology risk
- Flow alteration of 30 to 50 percent was assigned an orange node color representing high flow-ecology risk
- Flow alteration of 15 to 30 percent was assigned a yellow node color representing moderate flow-ecology risk
- Flow alteration of 0 to 15 percent was assigned a green node color representing low flow-ecology risk

For cottonwood recruitment the hydrologic metric is the same as for adult cottonwood and is also calculated for only wet years. The probability of cottonwood recruitment is calculated as:

- If flow alteration is 0 to -4 percent then recruitment = 1.
- If flow alteration is -4 to -55 percent then recruitment = 2.91 x %flow alteration^3 + 7.27 x %flow alteration^2 + 5.26 x %flow alteration + 1.21.

- If flow alteration -55 to -100 percent then recruitment = 0.

For cottonwood recruitment, the CDSS nodes were assigned different colors based on the following flow-ecology risk classes:

- Flow alteration of 30 to 100 percent was assigned a red node color representing very high flow-ecology risk
- Flow alteration of 18 to 30 percent was assigned an orange node color representing high flow-ecology risk
- Flow alteration of 7 to 18 percent was assigned a yellow node color representing moderate flow-ecology risk
- Flow alteration of 0 to 7 percent was assigned a green node color representing low flow-ecology risk

For cottonwood in confined settings the method developed in the pilot study was retained but applied only in moderate-energy confined geomorphic settings and at elevations less than 8,700 feet. The flow-ecology metric was calculated using the following equation:

$$\%\text{ departure from reference condition} = \frac{\text{Annual Peak Daily Flow}_{\text{current}} - \text{Annual Peak Daily Flow}_{\text{natural}}}{\text{Annual Peak Daily Flow}_{\text{natural}}} \times 100\%$$

For cottonwood in confined settings, the CDSS nodes were assigned different colors based on the following flow-ecology risk classes:

- Flow alteration of 42 to 100 percent was assigned a red node color representing very high flow-ecology risk
- Flow alteration of 21 to 42 percent was assigned an orange node color representing high flow-ecology risk
- Flow alteration of 8 to 21percent was assigned a yellow node color representing moderate flow-ecology risk
- Flow alteration of 0 to 8 percent was assigned a green node color representing low flow-ecology risk

2.7 Yampa River and White River Threatened and Endangered Fish Analysis Approach

Considerable work has been done through the Upper Colorado River Endangered Fish Recovery Program to understand the habitat and life-history needs of native, big-river, endangered fish as they relate to streamflow. These needs and relationships have been reported in peer-reviewed scientific publications, reports to the Recovery Program, management plans, and PBOs. The models developed for the WFET that were applied across the Yampa and White Basins do not apply to the endangered fish. To address the endangered fish, we consolidated and simplified Recovery Program information and recommendations in a manner that allowed us to compare current flows to recommended flows, as was done for other stream attributes.

2.7.1 Yampa River

Flow recommendations for endangered fish in designated critical habitat in the Yampa River (including Segment 1, Deerlodge Park, and Segment 13, Maybell) were derived from the PBO (USFWS 2005), documents referenced therein (e.g., Modde and Smith 1995, Modde et al. 1999), and additional guidance from USFWS on management of Elkhead Reservoir to augment baseflow needs at Maybell (USFWS 2008).

The PBO in combination with supporting documents (Modde and Smith 1995, Modde et al. 1999) indicates that the flow needs of endangered fish are met by flow remaining after the additional depletions described in the PBO. The future depletions specified in the PBO include 23,428 acre-feet (AF) above Lily and 30,104 AF above Maybell for a total of 53,532 AF above Deerlodge Park. StateMod was run using data representing the future depletions developed by CWCB. The resulting monthly flows were then used as the endangered fish flow recommendations against which current flows were compared.

Baseflow targets for July through March at Maybell were recommended by Modde et al. (1999) and were considered by the 2005 PBO in setting criteria for making storage releases from Elkhead Reservoir to augment base flows at the Maybell gage. These criteria for making storage releases were revised by USFWS in 2008. We analyzed how current flows could trigger Elkhead releases by calculating the proportion of days in each month on which the criteria for storage releases were met. The same approach was used for both Segment 13 (including Maybell) and Segment 1 (including Deerlodge Park). There are several points that should be kept in mind when reviewing results for Deerlodge Park: 1) the storage release criteria were developed for Maybell, not Deerlodge Park; if the criteria were developed for Deerlodge Park, they would likely be higher; 2) flows at Deerlodge Park include contributions from the Little Snake River, and 3) future depletions in Wyoming are likely not as well portrayed in StateMod as are depletions in Colorado. As interpreted for the WFET, we believe the criteria for storage releases to augment base flows at Maybell accurately represent USFWS recommendations but they cannot be considered a definitive statement of those recommendations.

2.7.2 White River

Draft flow recommendations for endangered fish on the White River were developed by the USFWS in 2011 (Mohrman 2011). These recommendations separately address spring peak flows and summer-winter baseflow, and flows in each of these periods are specified for five year types (wet, moderately wet, average, moderately dry, and dry). Spring peak flows are expressed as a flow rate and duration, e.g., for average years peak flows should be ≥ 2,900 cfs for 1 day and also remain ≥ 1,700 cfs for 25 days. Baseflows are expressed as a flow rate for July through March.

For inclusion in the WFET analysis, these recommendations were simplified in two ways. First, because StateMod models only monthly flows in the White River, all sub-monthly flow recommendations were converted into monthly values by combining sub-monthly recommendations with historic gage data. Where current monthly flows are less than recommended targets, historical flow patterns are not assumed to be augmented. Details on the method for converting the sub-monthly recommendations into monthly values are provided in the report on BOR's Colorado River Basin Study (USDOI BOR 2012). Second, because monthly flow recommendations across five different year types can be difficult to interpret, we averaged all year types into a single value for each month of the year, weighting year type by its expected frequency in the period of record.

The inclusion of the White River flow recommendations should be viewed as a means to understand these recommendations as they compare to current flows. This analysis presents a simplified view of these recommendations; it does not present a definitive statement on the status of flow compared to the recommendations.

2.8 Recreational Flow Relationship Approach

The purpose of the recreational analysis conducted as part of the study was to develop a natural set of information for whitewater recreation in the Yampa-White Basin. This information can be utilized in the future when evaluating future water management actions, climate change analyses, or risk management strategies. The following information was developed as part of the analysis:

- A list of whitewater recreation segments in the Yampa-White Basin.
- A survey collecting information from recreational users that developed recommended flow ranges for whitewater recreation in the Yampa-White Basin.
- A map showing the geographic extent of the whitewater recreation segments.
- A usable days analysis based on historic flow information and the flow ranges from the survey information. The analysis shows the average number of days in a given month that the reach would be usable based on flow information only. There are many factors that affect whether a whitewater recreation reach will be used on a given day beyond flow such as temperature, climatic conditions, financial considerations, permit availability, etc. The purpose of the analysis is to provide a natural set of data to provide insight into future water management decisions. The information can be one piece of information that is utilized in discussion of future water management activities in the basin.

The usable days approach includes instream flow survey data and the structural norm approach; a technique used to graphically represent social norms, and has been utilized to examine the acceptability of instream flows on river stretches across the United States and Canada for over 20 years (Whittaker & Shelby 2002). The graphic representation, commonly referred to as an impact acceptability curve, is used to describe optimum flows, ranges of tolerable flows, norm intensity, and level of norm agreement (Shelby, Vaske, &, Donnelly 1996). The Potential for Conflict Index (PCI) takes the graphic representation of social norms one step further by displaying information about their central tendency, dispersion, and form. Further details of these methods and results are presented in Appendix G.

2.9 Compact Deliveries and Nonconsumptive Needs Approach

The purpose of this analysis was to examine in general how nonconsumptive needs are supported by current river management to meet current water rights and the resulting compact deliveries. Using the flow-ecology relationships and flow-ecology risk mapping described above, areas that have low flow-ecology risk indicated environmental needs that are supported by current river management, water rights, and compact deliveries.

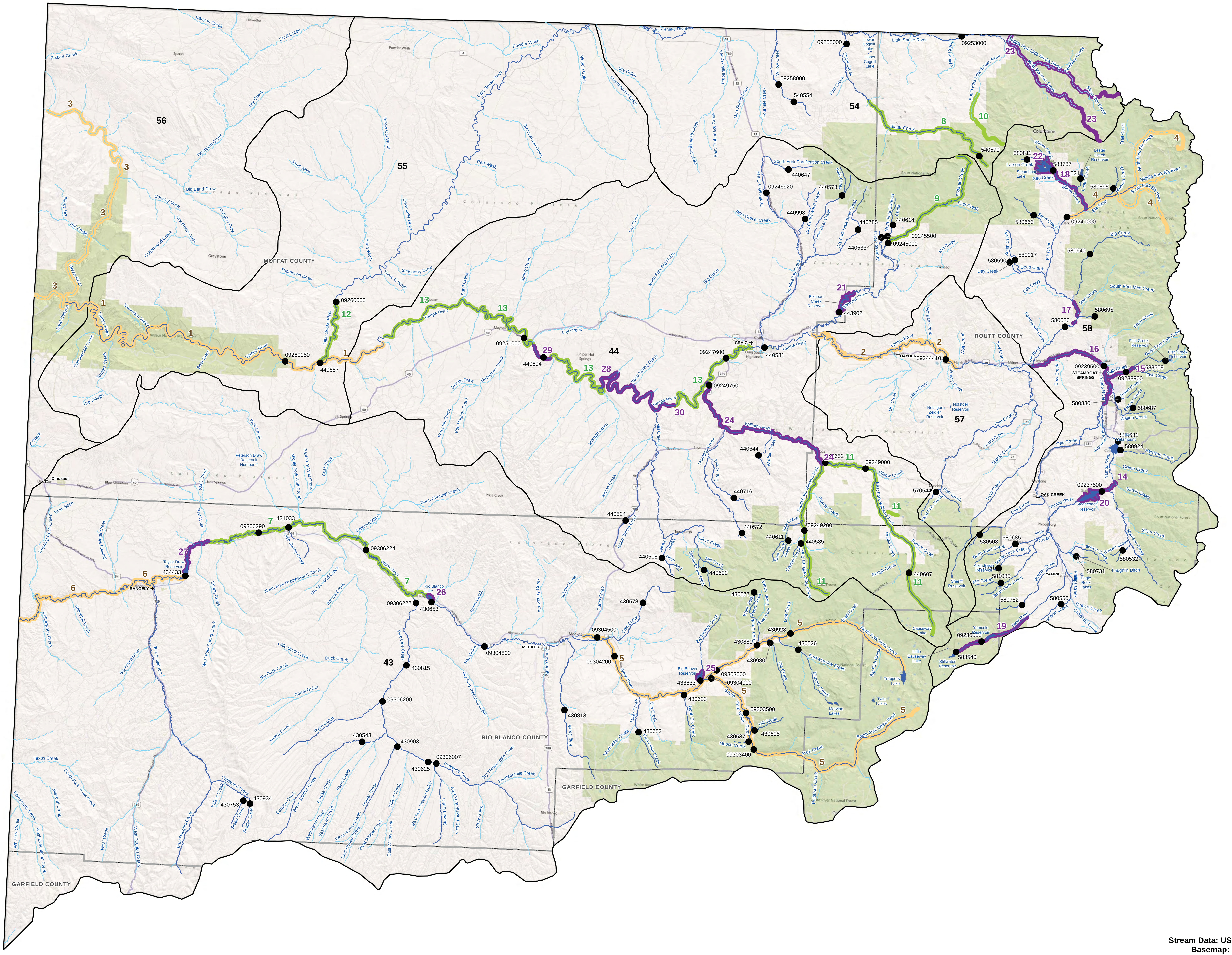


Figure 2-3

Yampa-White River Decision Support System StateMod Nodes with Natural and Existing Flow Conditions

Yampa - White Basin Watershed
Flow Evaluation Tool

- ● Diversion, Reservoir, and Instream Flow Node
- Study Stream
- Stream
- Environmental Segments
- Environmental and Recreational Segments
- Recreational Segments
- Highway
- Road
- + City and Town
- Yampa - White Basin Water District
- Lake and Reservoir
- County Boundary
- 🟩 National Forest / State Park

N S E W

0 2.5 5 10 15 20 Miles

CDM Smith

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Section 3

Watershed Flow Evaluation Tool Results

3.1 Watershed Flow Evaluation Tool Results Overview

This section summarizes the results of the flow-ecology risk mapping, recreation analysis, associated range of flow for the attributes previously mapped by the Yampa-White Basin, and the compact deliveries and nonconsumptive needs in the basin. Flow-ecology risk for trout, warm water fish, and riparian are summarized on color-coded maps (Figures 3-1 through 3-2) and in table format (Table 3-2). For the recreation analysis, information for the major recreation reaches is summarized using a usable days analysis. Finally, a summary of compact deliveries and nonconsumptive needs in the basin are described.

3.2 Flow-Ecology Mapping Results

In this section, the flow-ecology risk mapping results for trout, warm water fish, and riparian vegetation attributes are summarized. As discussed in Section 2, the flow-ecology metric for trout and warm water fish are based on low-flow metrics that occur in late summer and fall. For unconfined geomorphic settings, the riparian flow-ecology metric is based on a 90-day max flow metric that occurs during wet years and for confined settings, the flow-ecology metric is based on a one-day maximum over the full period of record.

Results for trout are included in Figures 3-1 and 3-2 at the end of this section. Nodes with lower risk of limiting trout populations are shown in green or blue. Nodes with higher risk of limiting trout populations are shown in red or orange. Figure 3-1 represents current water management conditions and for the White River, where all modeled locations are at low or minimal risk. For Yampa River current conditions, three of the locations examined have higher flow-ecology risk and the remaining 10 nodes are moderate to low risk. Locations with higher flow-ecology risk tend be located in tributaries higher in the watershed. Figure 3-2 shows results with future water management conditions and for the modeling conditions examined, results indicate no changes in flow-ecology risk between current and future conditions for the Yampa River.

Figures 3-1 and 3-2 at the end of this section also display the results for warm water flow-ecology risk mapping. Nodes with lower risk of reduced fish biomass are shown in green or blue and nodes with higher risk of reduced fish biomass are shown in red or orange. Figure 3-1 provides results for current water management conditions and Figure 3-2 represents future conditions for the Yampa River. For the White River current conditions, all DSS nodes examined are in the low flow-ecology risk category. For the Yampa River current conditions, results for upstream nodes indicate low to moderate flow-ecology risk and results for nodes on the lower Yampa and on the Little Snake River indicate high flow-ecology risk. When considering future demand conditions, one location (node 09251000) along the Yampa River has a higher risk category than current conditions as shown in Figure 3-2.

Figures 3-1 and 3-2 display the results for the riparian vegetation flow-ecology risk mapping, which includes an assessment of cottonwood in unconfined and confined settings for current and future conditions. Cottonwood flow-ecology risk in unconfined settings is based on the assessment of cottonwood recruitment and abundance. Nodes with lower flow-ecology risk are shown in green or yellow and nodes with higher flow-ecology risk are shown in orange or red. For current conditions, all locations in the Yampa and White Rivers have minimal or low flow-ecology risk and for future conditions, there is no change in flow-ecology risk for the Yampa River.

3.3 Recreational Flow Relationships Results

For recreation analysis, information was collected for the major recreation segments across the basin. These segments are listed in Table 3-1, and Figure 3-3 shows the geographic extent of each of the segments. Table 3-1 shows the segment mapped in Figure 3-3, the gage used to examine flows for recreational usable days analysis, and the range of flows for each segment derived from the American White Water Survey. Not all locations listed in Table 3-1 are suitable for a recreational usable days analysis due to lack of a gage for the segment or the amount of data points in the survey for a given reach were not sufficient for the analysis. The locations that were not included in the usable days analysis are listed here as they could potentially be improved upon in the future as more data becomes available.

Table 3-1 Flow Ranges for Whitewater Boating

Segment	Measurement Gage	Minimum (cfs)	Optimal (cfs)	Highest (cfs)
Fish Creek	USGS FISH CR AT UPPER STA NR STEAMBOAT SPRINGS, CO Gage 09238900	400	800-1,000	1,400
Steamboat Town	USGS YAMPA RIVER AT STEAMBOAT SPRINGS, CO Gage 09239500	700	1,500-2,700	5,000+
Elk River Box	USGS ELK RIVER NEAR MILNER, CO Gage 09242500	700	1,000-2,100	5,000+
Elk River - Clark	USGS ELK RIVER NEAR MILNER, CO Gage 09242500	700	1,300-4,000	5,000+
Willow Creek	DWR WILLOW CREEK, BELOW STEAMBOAT LAKE, CO Gage WILBSLCO	300	700-800	1,250
Mad Creek	Visual	400	400-1,000	2,000+
MF Little Snake	Visual	500	800-1,100	2,000+
Slater Creek	Insufficient survey data points to complete usable days analysis	600	1,100-2,100	3,000+
Yampa – Lower Town	USGS YAMPA RIVER ABOVE ELKHEAD CREEK NEAR HAYDEN, CO Gage 09244410	900	1,500	4,000
Little Yampa Canyon	USGS YAMPA RIVER BELOW CRAIG, CO Gage 09247600	1,100	1,700-2,500	10,000+
Cross Mountain Gorge	USGS YAMPA RIVER NEAR MAYBELL, CO Gage 09251000	700	1,500-3,500	5,000
Yampa Canyon	USGS YAMPA RIVER AT DEERLODGE PARK, CO Gage 09260050	1,300	2,700-20,000	20,000+
Gates of Lodore	USGS GREEN RIVER NEAR GREENDALE, UT Gage 09234500	1,100	1,900-15,000	20,000+
SF White River	Insufficient survey data points to complete usable days analysis	700	2,500-3,500	10,000
White River above Kenney Reservoir	Insufficient survey data points to complete usable days analysis	700	1,500-2,500	10,000+
White River Rangely to Bonanza	USGS WHITE RIVER BELOW BOISE CREEK, NEAR RANGELY, CO Gage 09306290	700	1,500-5,000	10,000+

Figures 3-4 through 3-14 show the results of the usable days analysis for the segments in Table 3-1. These charts show the average number of usable days per month for each segment for a given period of record. The period of record is indicated on each figure. For example, Figure 3-4 shows that for the month of May on average, 12 days are usable based on the minimum flow range and 15 days are usable based on the optimal flow range for a total of 27 days on average usable for May. The average daily flow is also included on each chart so that the annual hydrograph and usable days can be compared. The usable days analysis can be used to summarize the recreational season of each segment and the results shown in Figures 3-4 through 3-14 show that the months with the most usable days occur from April to May. The one exception is the Gates of Lodore segment (Figure 3-13), which has usable days in all months of the year.

3.4 Ranges of Flow Associated with Attributes Results

Table 3-2 on the following pages summarizes the ranges of flow associated with attributes previously mapped by the Yampa-White Basin Roundtable. The attributes were mapped by major environmental, recreational, or combined environmental/recreational segments based on input from the Yampa-White Basin Roundtable (CWCB 2011). For each segment, the attributes for each segment are listed. If a segment has whitewater recreation included, the range of flows presented in Table 3-1 above are summarized as well as the percentage of usable days by month based on results from Figures 3-4 through 3-14. For segments with warm water fish and/or trout, the range of flows associated with the flow-ecology risk levels are included for the months where the flow-ecology metrics is applied. If riparian attributes are present for a given segment, flow ranges based on the cottonwood recruitment flow-ecology metric are included for the months where the metric applies. Finally, for Threatened and Endangered Fish segments on the Yampa River, the PBO flow ranges are included and vary by month of the year. Similarly for the White River, flow ranges are included based on draft flow recommendations for endangered fish (Mohrman et al. 2011) as employed for the Colorado River Basin Water Supply and Demand Study (USDOI BOR 2012). The basin roundtable and its nonconsumptive committee requested this summary so that these results can be used in the future when examining future projects in the basin.

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 1 - Yampa River from entrance of Cross Mountain Canyon (East Cross Mountain) to confluence with Green River

Nodes:	09260050
Attributes:	whitewater boating, warm water fish, T&E fish, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Cross Mountain Gorge)[1]	minimal				700								
	optimal				1,500 to 3,500								
	highest				5,000								
	current condition				30% Days Usable	80% Days Usable	47% Days Usable	43% Days Usable	44% Days Usable				
Whitewater Boating (Yampa Canyon)[1]	minimal			1,300									
	optimal			2,700 to 20,000									
	highest			>20,000									
	current condition			27% Days Usable	86% Days Usable	99% Days Usable	92% Days Usable	43% Days Usable					
Warm Water Fish[2]	high							<30					
	moderate							30 to 120					
	recommended							120 to 215					
	low							>215					
	current condition							115					
	natural condition							305					
T&E Fish[3]	PBO flows (Elkhead trigger)	290 (124)	407 (124)	970 (---)	3,223 (---)	7,488 (---)	7,082 (---)	1,750 (120)	297 (120)	170 (120)	405 (120)	428 (124)	328 (124)
	Augmentation trigger met?	Met 99% of Days	Met 100% of Days	No recommended augmentation target				Met 90% of Days	Met 62% of Days	Met 47% of Days	Met 93% of Days	Met 96% of Days	Met 96% of Days
	current condition (1954 to 2005)	327	449	1018	3300	7600	7200	1798	323	189	449	465	363
Riparian[4]	very high				<7,200								
	high				7,200 to 8,500								
	moderate				8,500 to 9,600								
	low				>9,600								
	current condition				9,200								
	natural condition				10,300								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data; for Cross Mountain Gorge note that there is a maximum flow above which use of this reach does not typically occur (5,000 cfs) and therefore usable days are less than reaches with no maximum

[2] Warm water fish metric is based on 30-day low flow that occurs during July through November

[3] T&E fish low flow metrics from PBO and associated documents, including the USFWS 2008 Management of Water Releases from Elkhead etc.; peak flows are average flow that occurs April through June with 2045 depletions (53,000 AFY) included in CDSS model; current daily flows were examined and compared to PBO augmentation targets for statistics included in table

[4] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 2 - Yampa River from Pump Station to confluence of Elkhead Creek

Nodes: 09244410
Attributes: whitewater boating, trout, warm water fish, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Yampa River Lower Town)[1]	minimal				900								
	optimal				1,500 to 4,000								
	highest				4,000								
	current condition				59% Days Usable	65% Days Usable	41% Days Usable	31% Days Usable					
Trout[2]	very high								<110				
	high								110 to 170				
	moderate								170 to 280				
	minimal								280 to 615				
	low								>615				
	current condition								180				
	recommended								300				
Warm Water Fish[3]	high							<15					
	moderate							15 to 60					
	minimal							60 to 115					
	low							>115					
	current condition							110					
	natural condition							165					
Riparian[4]	very high				<3,400								
	high				3,400 to 4,000								
	moderate				4,000 to 4,600								
	low				>4,600								
	current condition				4,500								
	natural condition				4,900								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data; note that there is a maximum flow above which use of this reach does not typically occur (4,000 cfs) and therefore usable days are less than reaches with no maximum

[2] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[3] Warm water fish metric is based on 30-day low flow that occurs during July through November

[4] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 4 - Elk River from headwaters to the County Road 129 bridge at Clark; including the North, Middle and South Fork as well as the mainstem of the Elk

Nodes: 09241000
Attributes: whitewater boating, trout, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Elk River Box)[1]	minimal				700								
	optimal				1,000 to 2,100								
	highest				>5,000								
	current condition				36% Days Usable	96% Days Usable	88% Days Usable	26% Days Usable					
Whitewater Boating (Box Canyon to Clark)[1]	minimal				700								
	optimal				1,300 to 4,000								
	highest				>5,000								
	current condition				14% Days Usable	74% Days Usable	79% Days Usable	13% Days Usable					
Trout[2]	very high								<30				
	high								30 to 50				
	recommended								50 to 80				
	low								80 to 180				
	very low								>180				
	current condition								100				
	natural condition								110				
Riparian[3]	very high				<980								
	high				980 to 1,100								
	moderate				1,100 to 1,300								
	low				>1,300								
	current condition				1,400								
	natural condition				1,400								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data

[2] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 5 - White River from headwaters to Meeker; including the North and South Fork and mainstem of the White

Nodes: 09304500

Attributes: whitewater boating, trout, warm water fish, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Trout[1]	very high								<55				
	high								55 to 80				
	moderate								80 to 130				
	minimal								130 to 295				
	low								>295				
	current condition								350				
	natural condition								455				
Warm Water Fish[2]	high							<25					
	moderate							25 to 100					
	minimal							100 to 190					
	recommended							>190					
	current condition							250					
								265					
Riparian[3]	very high				<1,500								
	high				1,500 to 1,800								
	moderate				1,800 to 2,000								
	low				>2,000								
	current condition				1,900								
					2,100								

[1] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[2] Warm water fish metric is based on 30-day low flow that occurs during July through November

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 6 - White River below Kenney Reservoir dam to Utah State line

Nodes: 434433

Attributes: whitewater boating, trout, warm water fish, T&E fish, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (White River Rangely to Bonanza)[1]	minimal			700									
	optimal			1,500 to 5,000									
	highest			>10,000									
	current condition			10% Days Usable	43% Days Usable	88% Days Usable	80% Days Usable	37% Days Usable	14% Days Usable	8% Days Usable	14% Days Usable		
Warm Water Fish[2]	high							<30					
	moderate							30 to 125					
	minimal							125 to 230					
	low							>230					
	current condition							320					
	natural condition							320					
T&E Fish[3]	recommended	390	383	420	665	1495	1657	670	420	420	420	420	401
	current condition	390	383	501	655	1,686	1,979	834	470	433	452	450	401
Riparian[4]	very high				<1,700								
	high				1,700 to 2,000								
	moderate				2,000 to 2,200								
	low				>2,200								
	current condition				2,100								
	natural condition				2,400								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data

[2] Warm water fish metric is based on 30-day low flow that occurs during July through November

[3] T&E fish low average flow metrics based on Mohrman et al. 2011 Flow recommendations for the endangered fish of the White River, Colorado, and Utah (Draft), as interpreted by TNC in collaboration with USFWS and US Bureau of Reclamation for the Colorado River Basin Study (USDOI BOR 2012)

[4] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 7 - White River from Rio Blanco Lake to Kenney Reservoir

Nodes:	09306290
Attributes:	whitewater boating, warm water fish, T&E fish

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (White River above Kenney Reservoir)[1]	minimal				700								
	optimal				1,500 to 2,500								
	highest				>10,000								
	current condition				N/A	N/A	N/A	N/A					
Warm Water Fish[2]	very high							<30					
	high							30 to 120					
	moderate							120 to 220					
	low							>220					
	current condition							310					
	natural condition							310					
T&E Fish[3]	recommended												

[1] Flow ranges from American Whitewater Survey; usable days analysis not completed here due to lack of survey data

[2] Warm water fish metric is based on 30-day low flow that occurs during July through November

[3] Flow recommendations for endangered fish as specified only downstream of Kenny Reservoir; designated critical habitat for the endangered fish extends upstream to Rio Blanco Dam, but fish are blocked from upstream access by Taylor Draw Dam

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 8 - Slater Creek from headwaters to the Beaver Creek confluence

Nodes:	540570
Attributes:	whitewater boating, trout, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Slater Creek)[1]	minimal				600								
	optimal				1,100 to 2,100								
	highest				3,000								
	current condition				N/A	N/A	N/A	N/A					
Trout[2]	very high								<5				
	high								5 to 10				
	moderate								10 to 20				
	minimal								20 to 40				
	low								>40				
	current condition								10				
	recommended								10				
Riparian[3]	very high				<250								
	high				250 to 290								
	moderate				290 to 330								
	low				>330								
	current condition				350								
	natural condition				355								

[1] Flow ranges from American Whitewater Survey; usable days analysis not completed here due to lack of survey data

[2] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 9 - Elkhead Creek from headwaters to confluence of North Fork of Elkhead Creek

Nodes: 09245000

Attributes: trout, warm water fish, riparian

Attribute	Ecological Risk	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Trout[1]	very high								<5				
	high								5 to 10				
	moderate								10 to 15				
	minimal								15 to 30				
	low								>30				
	current condition								<5				
	natural condition								<5				
Warm Water Fish[2]	high												
	moderate												
	minimal												
	recommended												
	current condition												
	natural condition												
Riparian[3]	very high				<210								
	high				210 to 240								
	moderate				240 to 275								
	low				>275								
	current condition				295								
	natural condition				295								

[1] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[2] Warm water fish metric not applicable for this reach because flows are less than 5 cfs and there is uncertainty with modeled data less than 5 cfs

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 10 - South Fork of the Little Snake from headwaters to confluence of Johnson Creek

Nodes:	09253000
Attributes:	trout, warm water fish, riparian

Attribute	Ecological Risk	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Trout[1]	very high								<25				
	high								25 to 35				
	moderate								35 to 60				
	minimal								60 to 135				
	low								>135				
	current condition								30				
	natural condition								45				
Warm Water Fish[2]	high							<3					
	moderate							3 to 11					
	minimal							11 to 20					
	recommended							>20					
	current condition							20					
	natural condition							30					
Riparian[3]	very high				<820								
	high				820 to 960								
	moderate				960 to 1,100								
	low				>1,100								
	current condition				1,100								
	natural condition				1,200								

[1] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[2] Warm water fish metric is not applicable for this reach because flows are less than 5 cfs and there is uncertainty with modeled data less than 5 cfs

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 11 - South and East Fork of the Williams Fork from headwaters to the confluence of the Forks and Segment 24 - Williams Fork - from South Fork to confluence of the Yampa River

Nodes: 09249000, 09249200, 09249750

Attributes: trout, riparian

Attribute	Ecological Risk	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Trout (East Fork)[1] 09249000	very high								<12				
	high								12 to 18				
	moderate								18 to 30				
	minimal								30 to 65				
	low								>65				
	current condition								40				
	natural condition								50				
Trout (South Fork)[1] 09249200	very high								<4				
	high								4 to 6				
	moderate								6 to 10				
	recommended								10 to 20				
	low								>20				
	current condition								5				
	natural condition								5				
Trout[1] 09249750	very high								<25				
	high								25 to 35				
	moderate								35 to 60				
	minimal								60 to 130				
	low								>130				
	current condition								45				
	natural condition								70				
Riparian[2] 09249750	very high				<710								
	high				710 to 830								
	moderate				830 to 940								
	low				>940								
	current condition				940								
	natural condition				1,000								

[1] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[2] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 12 - Little Snake River from Moffat County Road 10 to confluence of the Yampa River

Nodes: 09260000

Attributes: warm water fish, riparian

Attribute	Ecological Risk	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Warm Water Fish[1]	high							<5					
	moderate							5 to 15					
	minimal							15 to 30					
	low							>30					
	current condition							10					
	natural condition							40					
Riparian[2]	very high				<2,100								
	high				2,100 to 2,400								
	moderate				2,400 to 2,800								
	low				>2,800								
	recommended				2,600								
	natural condition				3,000								

[1] Warm water fish metric is based on 30-day low flow that occurs during July through November

[2] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 13 - Yampa River from Craig (Hwy 394 Bridge) to mouth of Cross Mountain Canyon

Nodes: 09251000

Attributes: whitewater boating, trout, warm water fish, riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Little Yampa Canyon)[1]	minimal			1100									
	optimal			1,700 to 2,500									
	highest			>10,000									
	current condition			10% Days Usable	71% Days Usable	100% Days Usable	90% Days Usable	33% Days Usable	14% Days Usable				
Warm Water Fish[2]	high							<25					
	moderate							25 to 90					
	minimal							90 to 170					
	low							>170					
	current condition							100					
	natural condition							240					
T&E Fish[3]	recommended	231 (124)	302 (124)	667 (---)	2,369 (---)	5,796 (---)	5,264 (---)	1,287 (120)	226 (120)	140 (120)	291 (120)	302 (124)	242 (124)
	Augmentation trigger met?	Met 98% of Days	Met 98% of Days	No recommended augmentation target				Met 81% of Days	Met 55% of Days	Met 36% of Days	Met 93% of Days	Met 95% of Days	Met 89% of Days
	current condition (1954 to 2005)	260	334	706	2,411	5,833	5,305	1,326	249	155	327	328	268
Riparian[4]	very high				<5,200								
	high				5,200 to 6,100								
	moderate				6,100 to 7,000								
	low				>7,000								
	current condition				6,800								
	natural condition				7,500								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data

[2] Warm water fish metric is based on 30-day low flow that occurs during July through November

[3] T&E fish low flow metrics from PBO and associated documents, including the USFWS 2008 Management of Water Releases from Elkhead etc.; peak flows are average flow that occurs April through June with 2045 depletions (53,000 AFY) included in CDSS model; historic daily flows were examined and compared to PBO augmentation targets for statistics included in table

[4] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 14 - Yampa River from Stagecoach Reservoir "Tailwaters" to northern boundary of Sarvis Creek State Wildlife area

Nodes: 09237500
Attributes: trout, warm water fish, riparian

<table>
<tr><th>Attribute</th><th>Ecological Risk or Recreational Flow Preference</th><th>Jan (cfs)</th><th>Feb (cfs)</th><th>Mar (cfs)</th><th>Apr (cfs)</th><th>May (cfs)</th><th>Jun (cfs)</th><th>Jul (cfs)</th><th>Aug (cfs)</th><th>Sep (cfs)</th><th>Oct (cfs)</th><th>Nov (cfs)</th><th>Dec (cfs)</th></tr>
<tr><td rowspan="7">Trout[1]</td><td>very high</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2"><10</td><td></td><td></td><td></td></tr>
<tr><td>high</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">10 to 20</td><td></td><td></td><td></td></tr>
<tr><td>moderate</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">20 to 30</td><td></td><td></td><td></td></tr>
<tr><td>minimal</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">30 to 70</td><td></td><td></td><td></td></tr>
<tr><td>low</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="2">>70</td><td></td><td></td><td></td></tr>
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<tr><td rowspan="6">Warm Water Fish[2]</td><td>high</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5"><5</td><td></td></tr>
<tr><td>moderate</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">5 to 20</td><td></td></tr>
<tr><td>minimal</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">20 to 35</td><td></td></tr>
<tr><td>recommended</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">>35</td><td></td></tr>
<tr><td>current condition</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">55</td><td></td></tr>
<tr><td>natural condition</td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="5">50</td><td></td></tr>
<tr><td rowspan="6">Riparian[3]</td><td>very high</td><td></td><td></td><td></td><td colspan="3"><280</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>high</td><td></td><td></td><td></td><td colspan="3">280 to 340</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>moderate</td><td></td><td></td><td></td><td colspan="3">340 to 380</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>low</td><td></td><td></td><td></td><td colspan="3">>380</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>current condition</td><td></td><td></td><td></td><td colspan="3">235</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>natural condition</td><td></td><td></td><td></td><td colspan="3">400</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

[1] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[2] Warm water fish metric is based on 30-day low flow that occurs during July through November

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 15 - Fish Creek from Fish Creek Falls to confluence of the Yampa River

Nodes: 09238900
Attributes: whitewater boating and riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Fish Creek)[1]	minimal					400							
	optimal					800 to 1,000							
	highest					1,400							
	current condition					11% Days Usable	42% Days Usable						
Riparian[2]	very high				<210								
	high				210 to 250								
	moderate				250 to 280								
	low				>280								
	current condition				290								
	natural condition				300								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data recommended

[2] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 16 - Yampa River from Chuck Lewis Wildlife Area to Pump Station

Nodes: 09239500

Attributes: whitewater boating, trout, warm water fish and riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Steamboat Town)[1]	minimal				700								
	optimal				1,500 to 2,700								
	highest				>5,000								
	current condition				39% Days Usable	87% Days Usable	78% Days Usable	13% Days Usable					
Whitewater Boating (Lower Town Run)[1]	minimal				900								
	optimal				1,500								
	highest				4,000								
	current condition				59% Days Usable	65% Days Usable	41% Days Usable	31% Days Usable					
Trout[2]	very high								<50				
	high								50 to 75				
	recommended								75 to 125				
	minimal								125 to 280				
	low								>280				
	current condition								115				
	natural condition								150				
Warm Water Fish[3]	high							<10					
	moderate							10 to 35					
	minimal							35 to 60					
	low							>60					
	current condition							85					
	natural condition							90					
Riparian[4]	very high				<1,510								
	high				1,510 to 1,770								
	moderate				1,770 to 2,005								
	low				>2,005								
	current condition				1,900								
	natural condition				2,200								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data

[2] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[3] Warm water fish metric is based on 30-day low flow that occurs during July through November

[4] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 18 - Willow Creek below Steamboat Lake to confluence with the Elk

Nodes:	583787
Attributes:	whitewater boating, trout, and riparian

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Whitewater Boating (Willow Creek)[1]	minimal					300							
	optimal					700 to 800							
	highest					1,250							
	current condition					9% Days Usable	5% Days Usable						
Trout[2]	very high								<4				
	high								4 to 6				
	moderate								6 to 10				
	minimal								10 to 25				
	low								>25				
	current condition								15				
	recommended								15				
Riparian[3]	very high				<125								
	high				125 to 145								
	moderate				145 to 165								
	low				>165								
	current condition				175								
	natural condition				180								

[1] Flow ranges from American Whitewater Survey; usable days compares flow ranges with historical data

[2] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

[3] Riparian metrics are based on peak flows that occur on average 3 out of 10 years; 90-day metric; recruitment metric included here as most conservative

Table 3-2 Results of StateMod Modeling of Current Conditions Compared to Flow Ranges for Nonconsumptive Attributes

Segment 19 - Bear River from headwaters to USFS boundary

Nodes: 09236000
Attributes: trout

Attribute	Ecological Risk or Recreational Flow Preference	Jan (cfs)	Feb (cfs)	Mar (cfs)	Apr (cfs)	May (cfs)	Jun (cfs)	Jul (cfs)	Aug (cfs)	Sep (cfs)	Oct (cfs)	Nov (cfs)	Dec (cfs)
Trout[1]	very high								<4				
	high								4 to 6				
	moderate								6 to 10				
	minimal								10 to 20				
	low								>20				
	current condition								40				
	natural condition								30				

[1] Trout metric is based on average of August and September mean flow divided by mean annual natural flow

3.5 Compact Deliveries and Nonconsumptive Needs Results

Based on results discussed in Section 3.3, trout, warm water fish, and riparian attributes in the White River Basin, in general, have low to moderate flow-ecology risk. Therefore, for the White River Basin, it can be assumed that current river management and water rights, including deliveries out of state from the Yampa River that are credited as compact deliveries, are supportive of nonconsumptive needs during both low flow and high flow conditions.

For the Yampa River Basin, most locations across the Yampa River Basin have trout, warm-water fish, and riparian attributes with low to moderate flow-ecology risk. However, there are several locations where high flow-ecology risk is indicated and a few locations where very high flow-ecology risk is indicated. The one consistent pattern that emerges from these higher risk locations is that native warm-water fish in the lower part of the river are at-risk due to late season low flows. This finding is consistent with findings and actions in the Upper Colorado River Basin Endangered Fish Recovery Program, which has augmented baseflows for the endangered fish through the enlargement and management of Elkhead Reservoir. Low flow conditions in the lower reaches of the basin, even under augmented conditions, still pose some risk to warm water and endangered fish. In summary, for the Yampa River Basin, it can be assumed that current river and water rights are generally supportive of nonconsumptive needs, but there is one known cause for concern in the lower basin and a few locations higher in the basin where risk may be indicated.

Notwithstanding late summer low flow concerns on the Yampa River, flows delivered from the Yampa and White that are credited to compact delivery, are highly supportive of the endangered fish of the Upper Colorado Basin. Tyus and Saunders (2001) noted that "the Yampa River is the most important tributary for recovering the endangered fishes" and they also noted that the White River is important for the endangered fish. The value of these rivers also extends onto the Green River. In both cases, the high value of the river is described by Tyus and Saunders as being in large part due to relatively unmanaged flows. The Yampa River PBO states that flows currently provided by the Yampa River are important for creating and maintaining endangered fish habitats on the Green River (USFWS 2005). Mohrman et al. (2011) reached similar conclusions for the White River.

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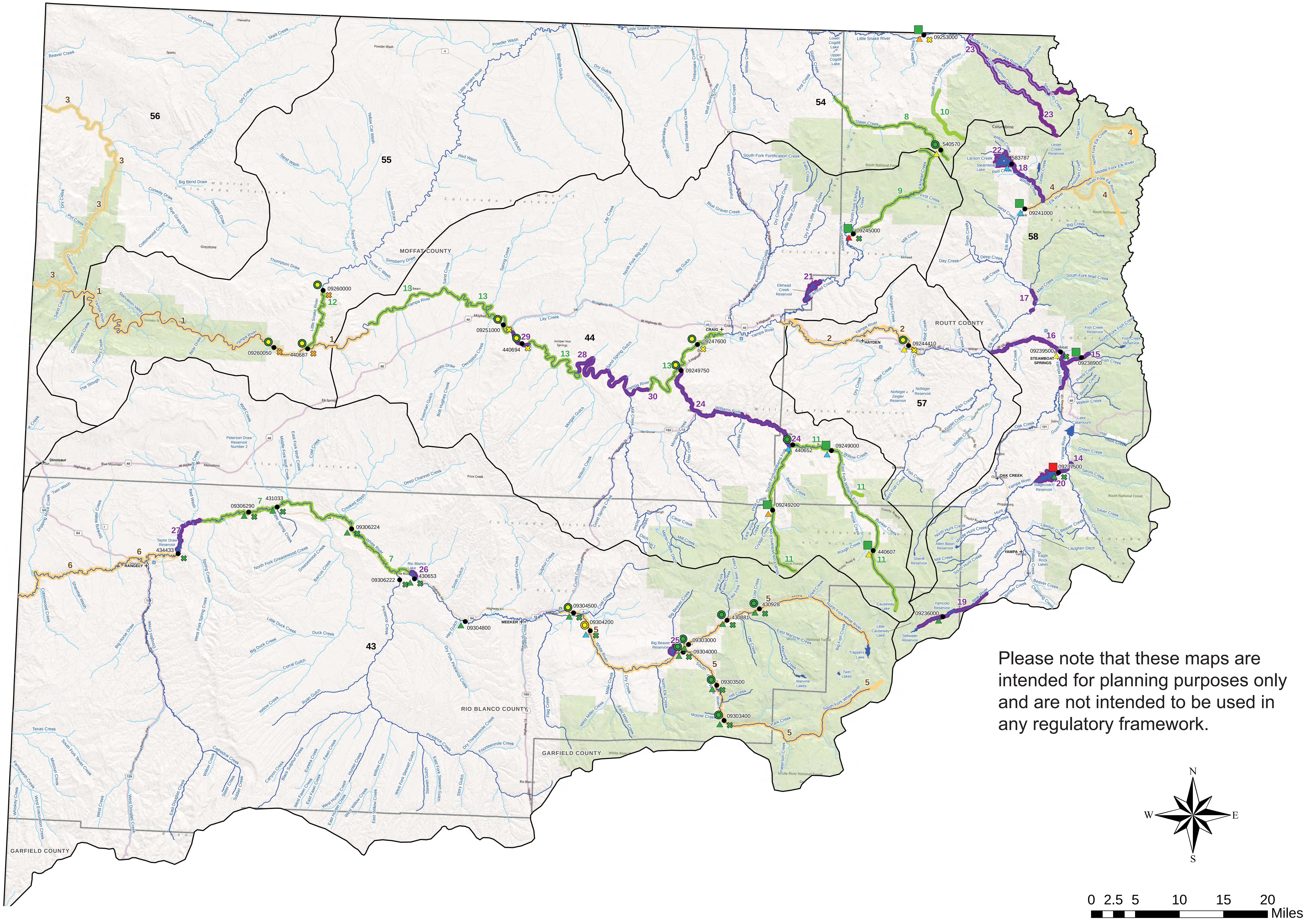
Stream Data: USGS National Hydrography Dataset
Basemap: Microsoft Bing Maps Web Service



Please note that these maps are intended for planning purposes only and are not intended to be used in any regulatory framework.

Cottonwood in Unconfined Settings

Recruitment

- ● -30% to -100% Very High Risk
- ● -18% to -30% High Risk
- ● -7% to -18% Moderate Risk
- ● 0% to -7% Low Risk

Abundance

- ● -50% to -100% Very High Risk
- ● -30% to -50% High Risk
- ● -15% to -30% Moderate Risk
- ● 0% to -15% Low Risk

Cottonwood in Confined Settings

- 🟥 -42% to -100% Very High Risk
- 🟧 -21% to -42% High Risk
- 🟨 -8% to -21% Moderate Risk
- 🟩 0% to -8% Low Risk

Warm Water Fish Flow-Ecology Risk

% Change in Biomass

- ✖ > 50% Very High Risk
- ✖ 25% to 50% High Risk
- ✖ 10% to 25% Moderate Risk
- ✖ < 10 % Low Risk

Trout Flow-Ecology Risk

- ▲ <10% Inadequate to support trout (Very High Risk)
- ▲ 10% to 15% Inadequate to support trout (High Risk)
- ▲ 16% to 25% May severely limit trout stock every few years (Moderate Risk)
- ▲ 26% to 55% Low flow may occasionally limit trout numbers (Minimal Risk)
- ▲ >55% Low flow may very seldom limit trout (Low Risk)

- ● CDSS Model Nodes
- Environmental Segments
- Environmental and Recreational Segments
- Recreational Segments
- Study Stream
- Stream
- \+ City and Town
- Yampa - White Basin Water District
- Lake and Reservoir
- County Boundary
- National Forest / State Park

Figure 3-1.
Riparian, Warm Water Fish, and Trout Flow-Ecology Risk Mapping (Current Conditions)

Yampa - White Basin Watershed
Flow Evaluation Tool

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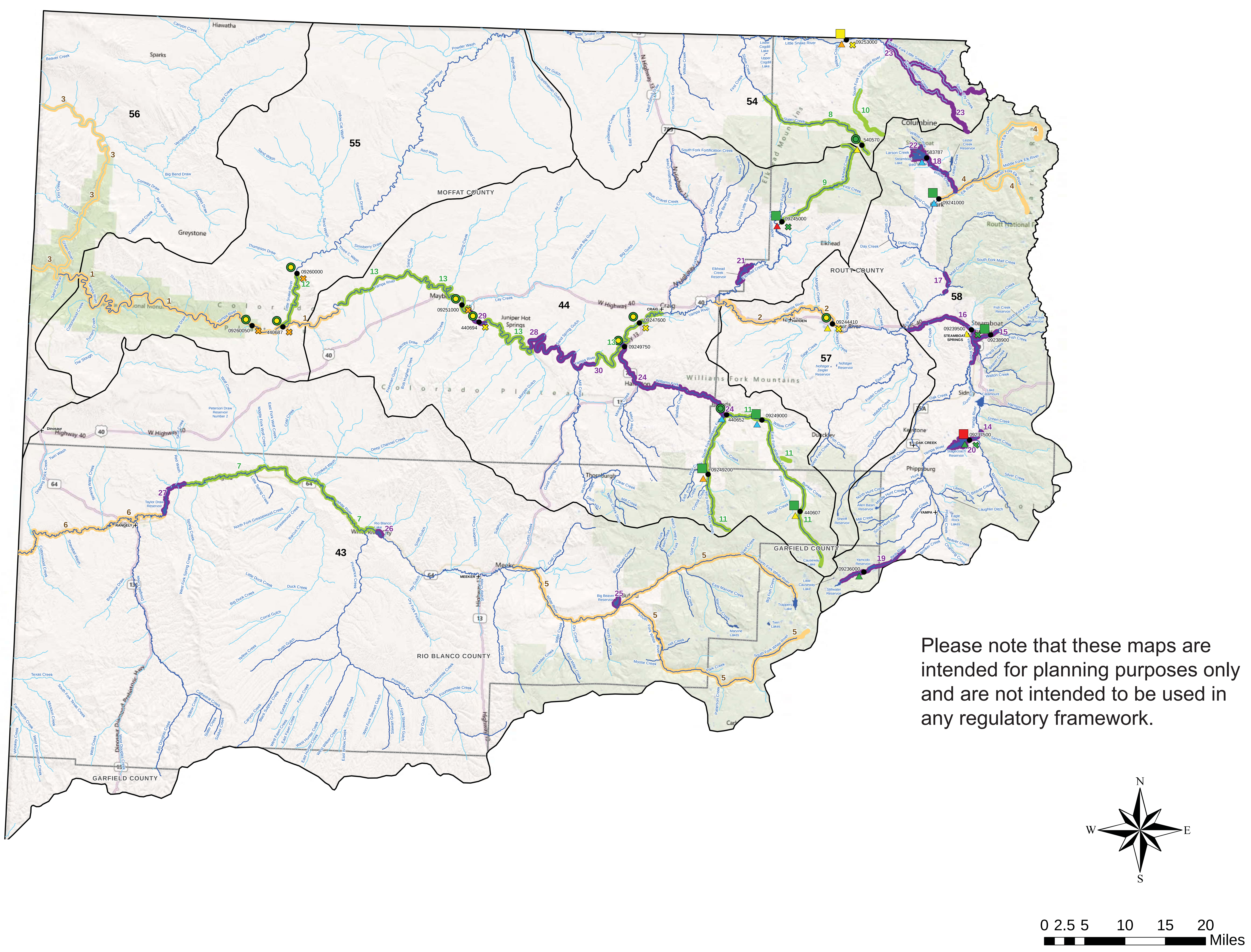
Stream Data: USGS National Hydrography Dataset
Basemap: Microsoft Bing Maps Web Service



Please note that these maps are intended for planning purposes only and are not intended to be used in any regulatory framework.

Figure 3-2.
Riparian, Warm Water Fish, and Trout Flow-Ecology Risk Mapping (Future Conditions)

Yampa - White Basin Watershed
Flow Evaluation Tool

Cottonwood in Unconfined Settings

Recruitment
- ● -30% to -100% Very High Risk
- ● -18% to -30% High Risk
- ● -7% to -18% Moderate Risk
- ● 0% to -7% Low Risk

Abundance
- ● -50% to -100% Very High Risk
- ● -30% to -50% High Risk
- ● -15% to -30% Moderate Risk
- ● 0% to -15% Low Risk

Cottonwood in Confined Settings
- 🟥 -42% to -100% Very High Risk
- 🟧 -21% to -42% High Risk
- 🟨 -8% to -21% Moderate Risk
- 🟩 0% to -8% Low Risk

Warm Water Fish Flow-Ecology Risk

% Change in Biomass
- ✖ > 50% Very High Risk
- ✖ 25% to 50% High Risk
- ✖ 10% to 25% Moderate Risk
- ✖ < 10 % Low Risk

Trout Flow-Ecology Risk
- ▲ <10% Inadequate to support trout (Very High Risk)
- ▲ 10% to 15% Inadequate to support trout (High Risk)
- ▲ 16% to 25% May severely limit trout stock every few years (Moderate Risk)
- ▲ 26% to 55% Low flow may occasionally limit trout numbers (Minimal Risk)
- ▲ >55% Low flow may very seldom limit trout (Low Risk)

- ● CDSS Model Nodes
- Environmental Segments
- Environmental and Recreational Segments
- Recreational Segments
- Study Stream
- Stream
- + City and Town
- Yampa - White Basin Water District
- Lake and Reservoir
- County Boundary
- National Forest / State Park

CDM Smith

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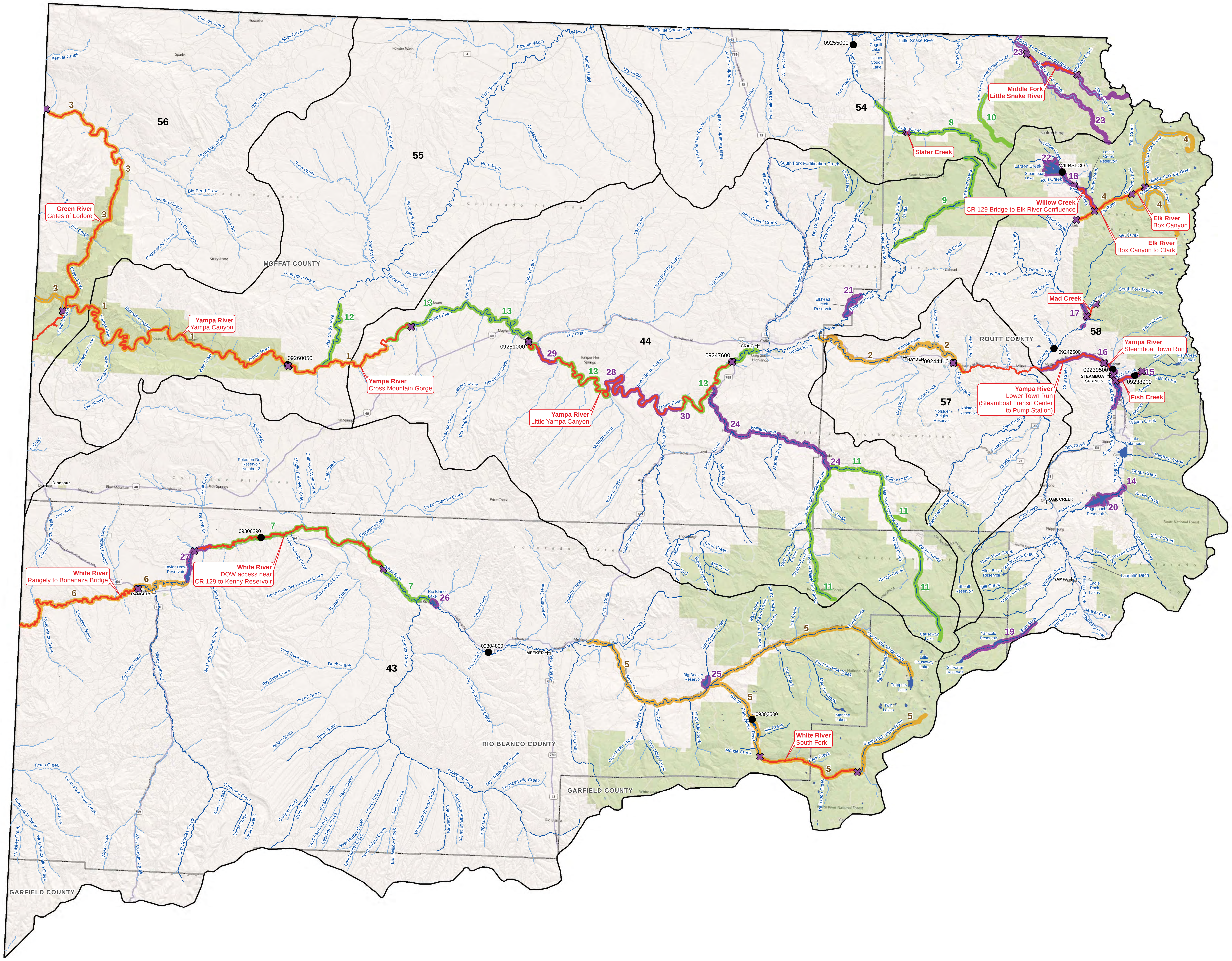

Stream Data: USGS National Hydrography Dataset
Basemap: Microsoft Bing Maps Web Service

Figure 3-3
Yampa-White Basin Whitewater Reaches and CDSS Nodes Used in Useable Days Analysis

Yampa-White Basin Watershed
Flow Evaluation Tool

- ● Gauge Nodes
- ✖ Recreational Segment Start / Stop
- Recreational Segment
- Study Stream
- Stream

Focus Area Segment

- Environmental
- Environmental & Recreational
- Recreational
- National Forest / State Park

- + City and Town
- Highway
- Road
- Lake and Reservoir
- Yampa-White Basin Water District
- County Boundary

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CDM Smith

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Figure 3-4 Fish Creek
Mean Number of Usable Days by Month (1954-2005)

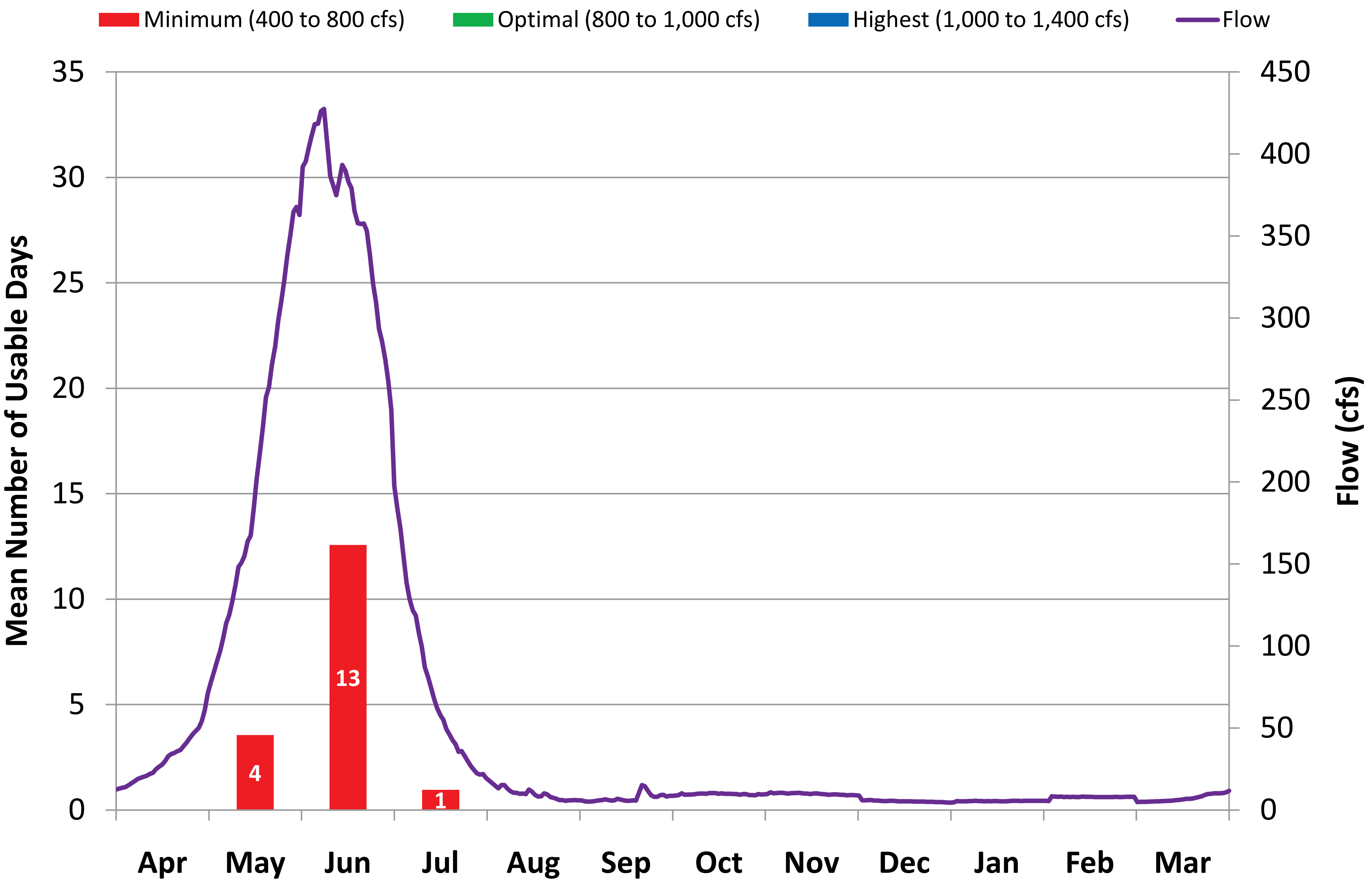


Figure 3-5 Steamboat Town
Mean Number of Usable Days by Month (1954-2005)

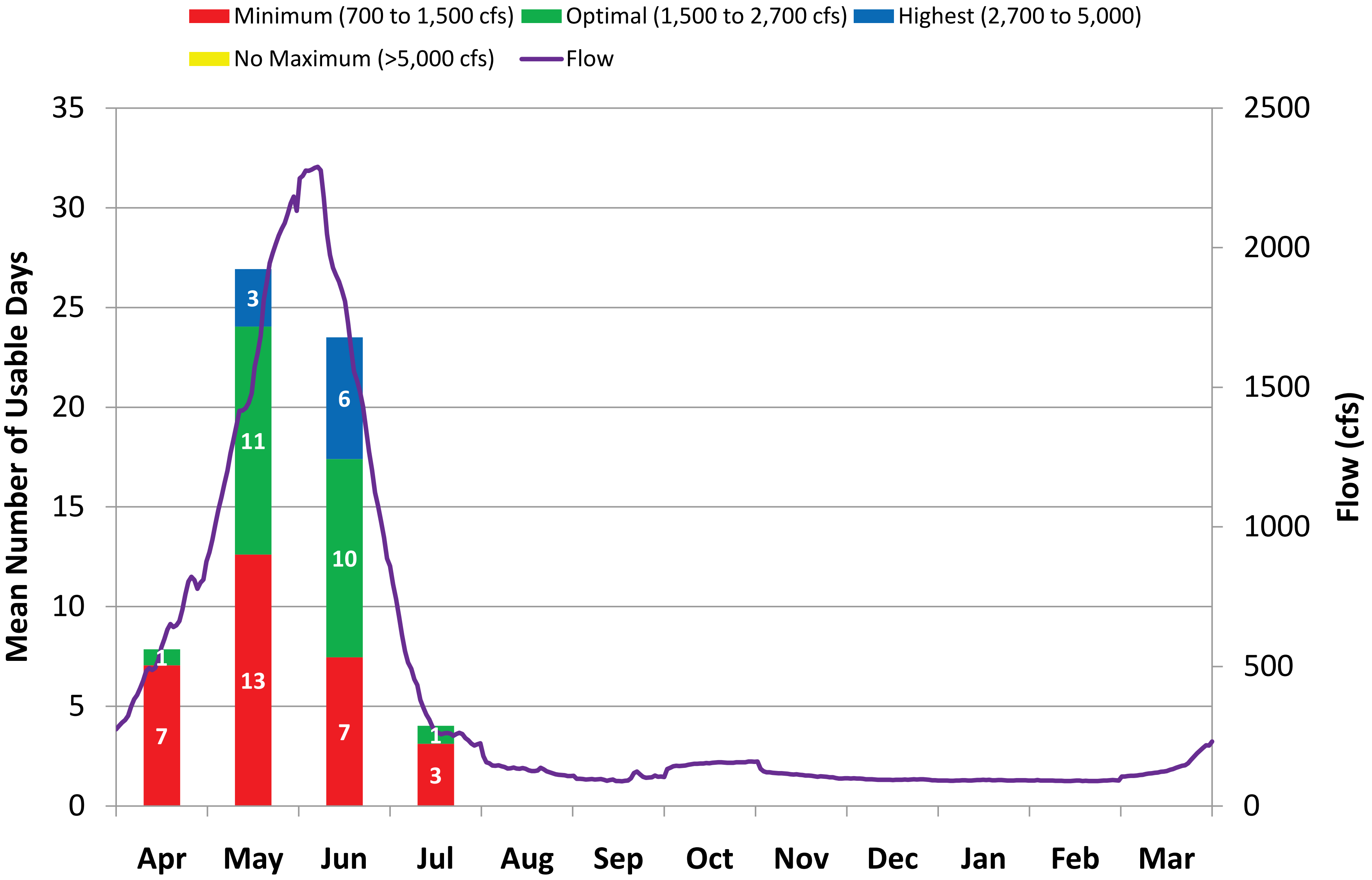


Figure 3-6 Elk River Box
Mean Number of Usable Days by Month (1991-2010)

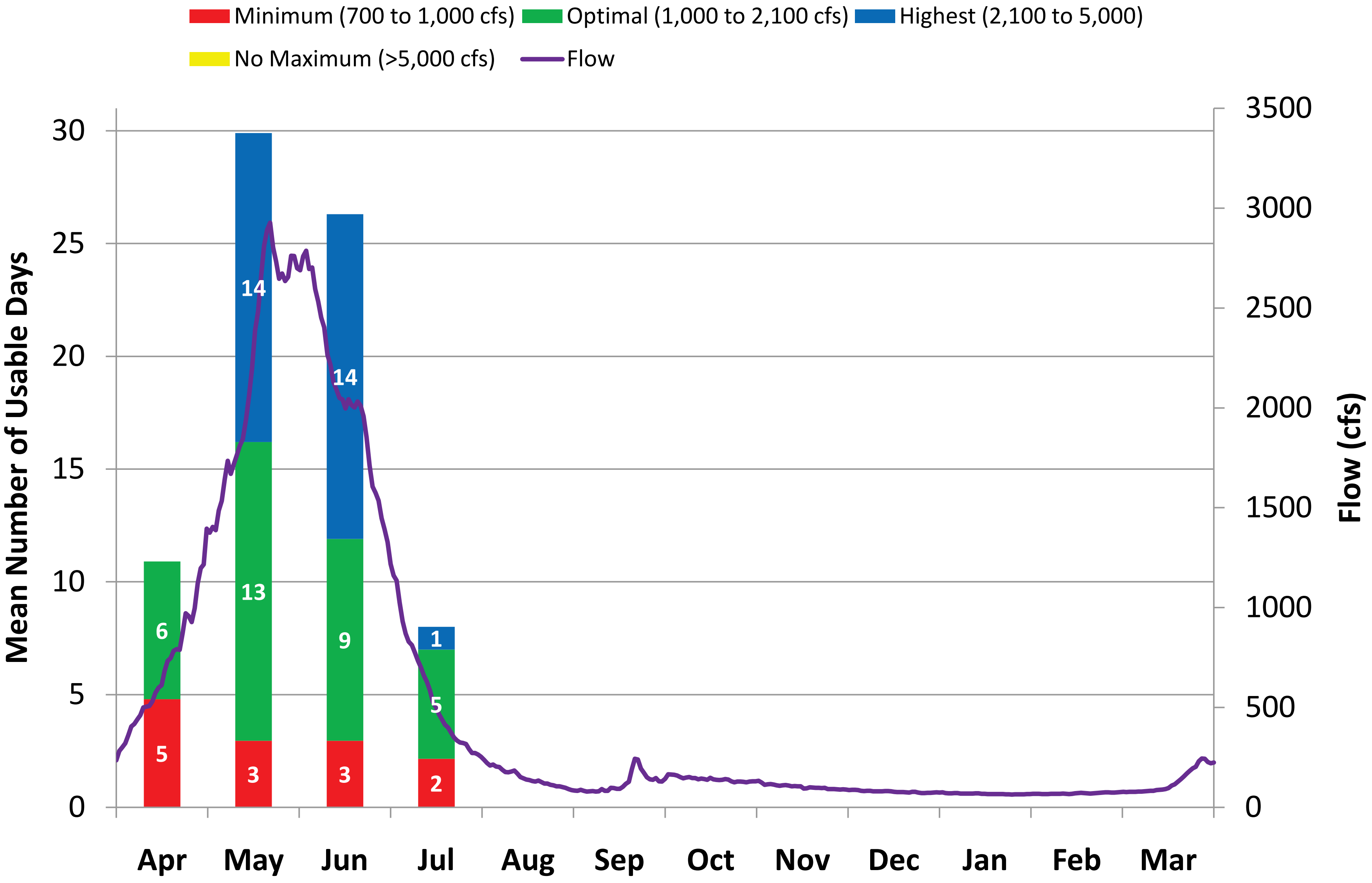


Figure 3-7 Elk River - Clark
Mean Number of Usable Days by Month (1991-2010)

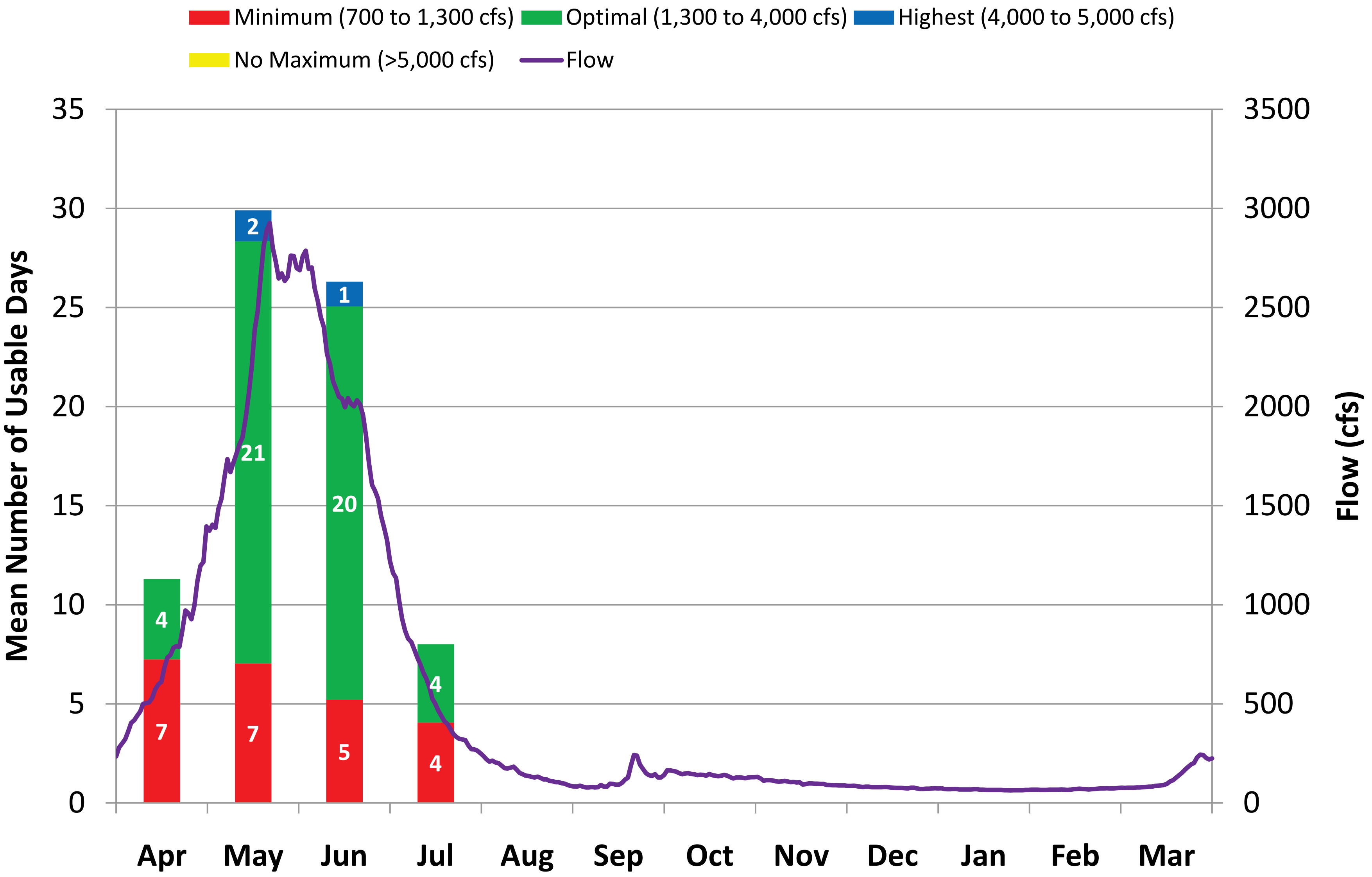


Figure 3-8 Willow Creek

Mean Number of Usable Days by Month (1979-2010)

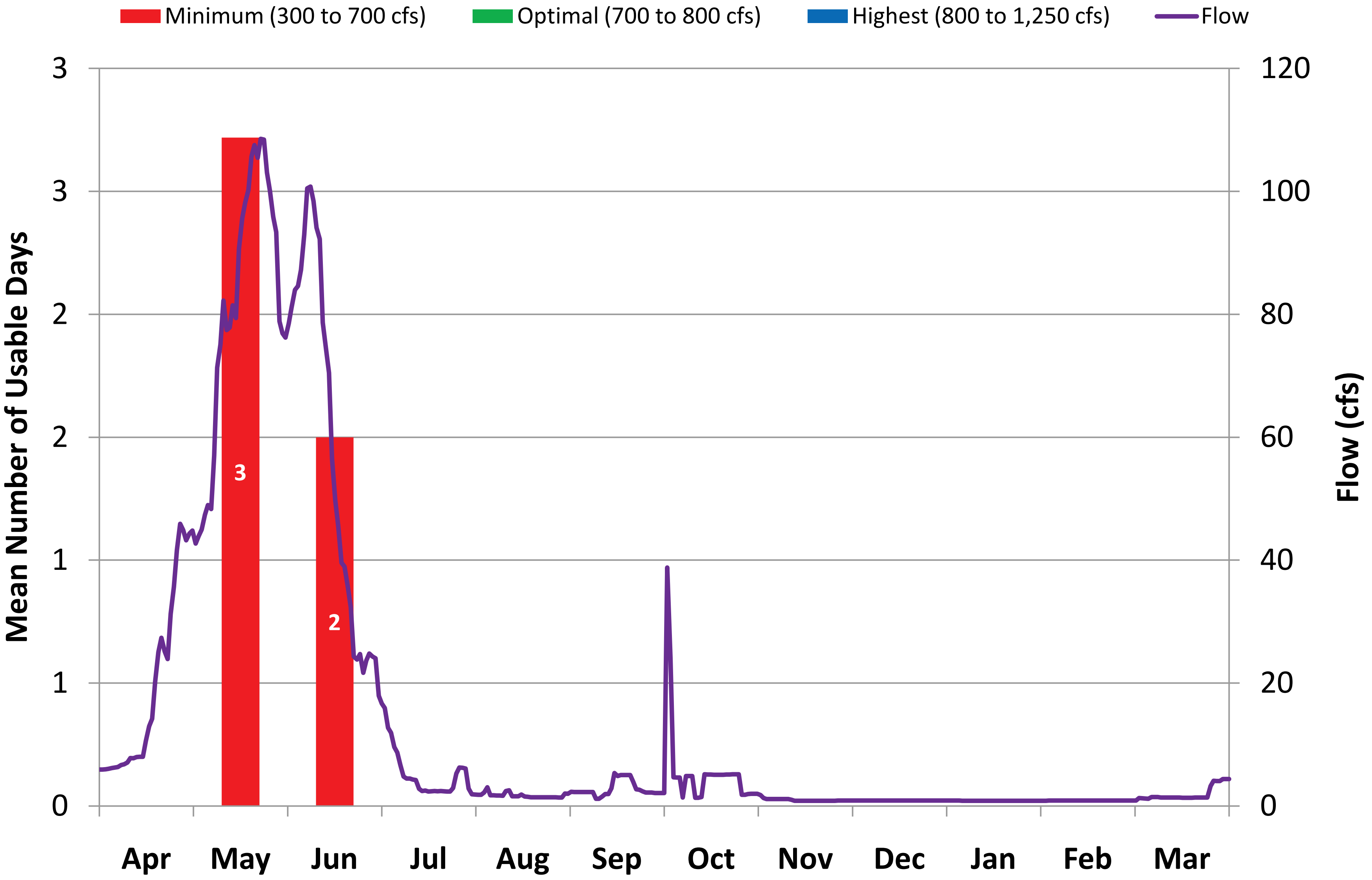


Figure 3-9 Yampa - Lower Town

Mean Number of Usable Days by Month (1954-2005)

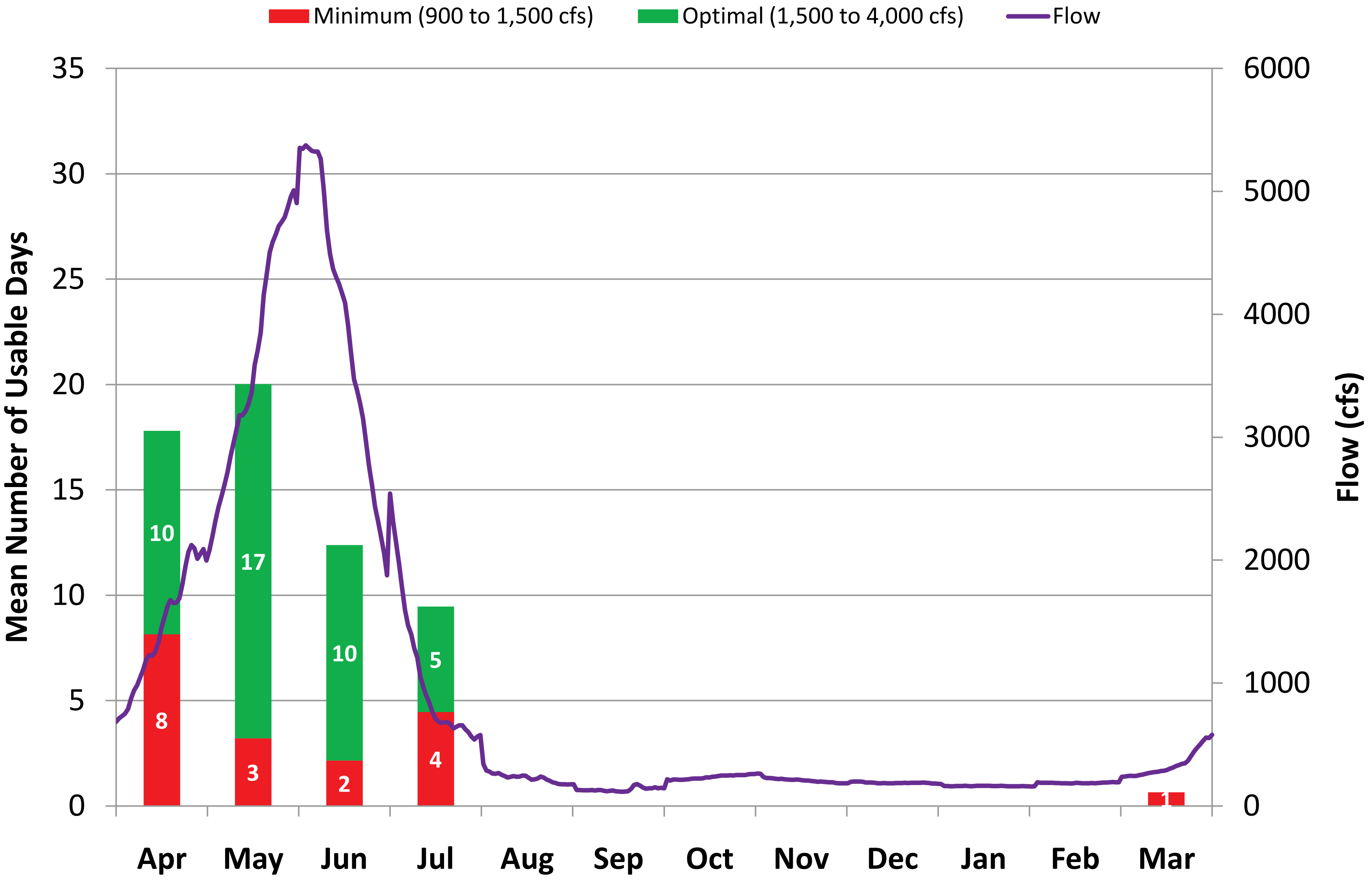


Figure 3-10 Little Yampa Canyon
Mean Number of Usable Days by Month (1954-2005)

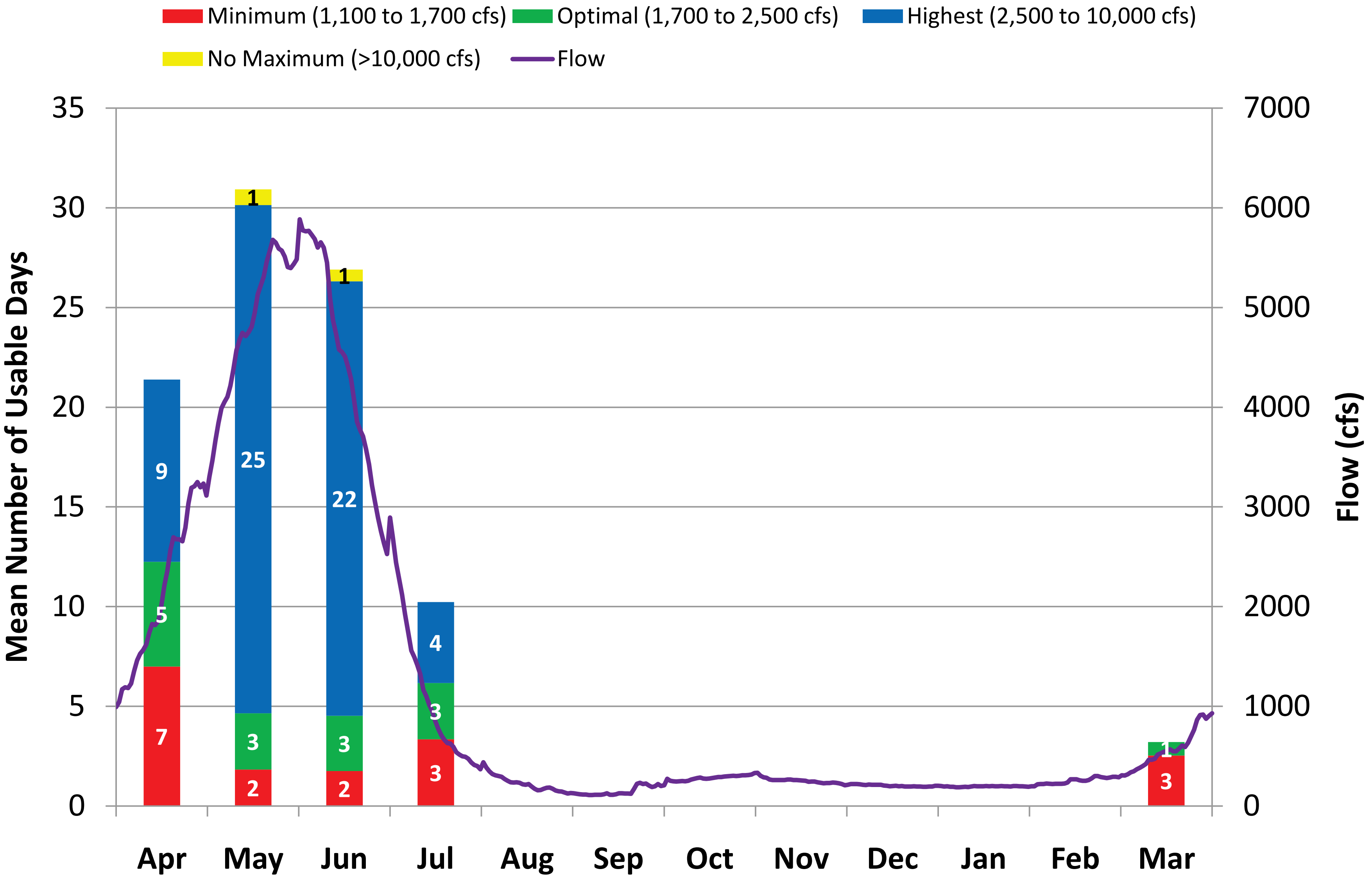


Figure 3-11 Cross Mountain Gorge

Mean Number of Usable Days by Month (1954-2005)

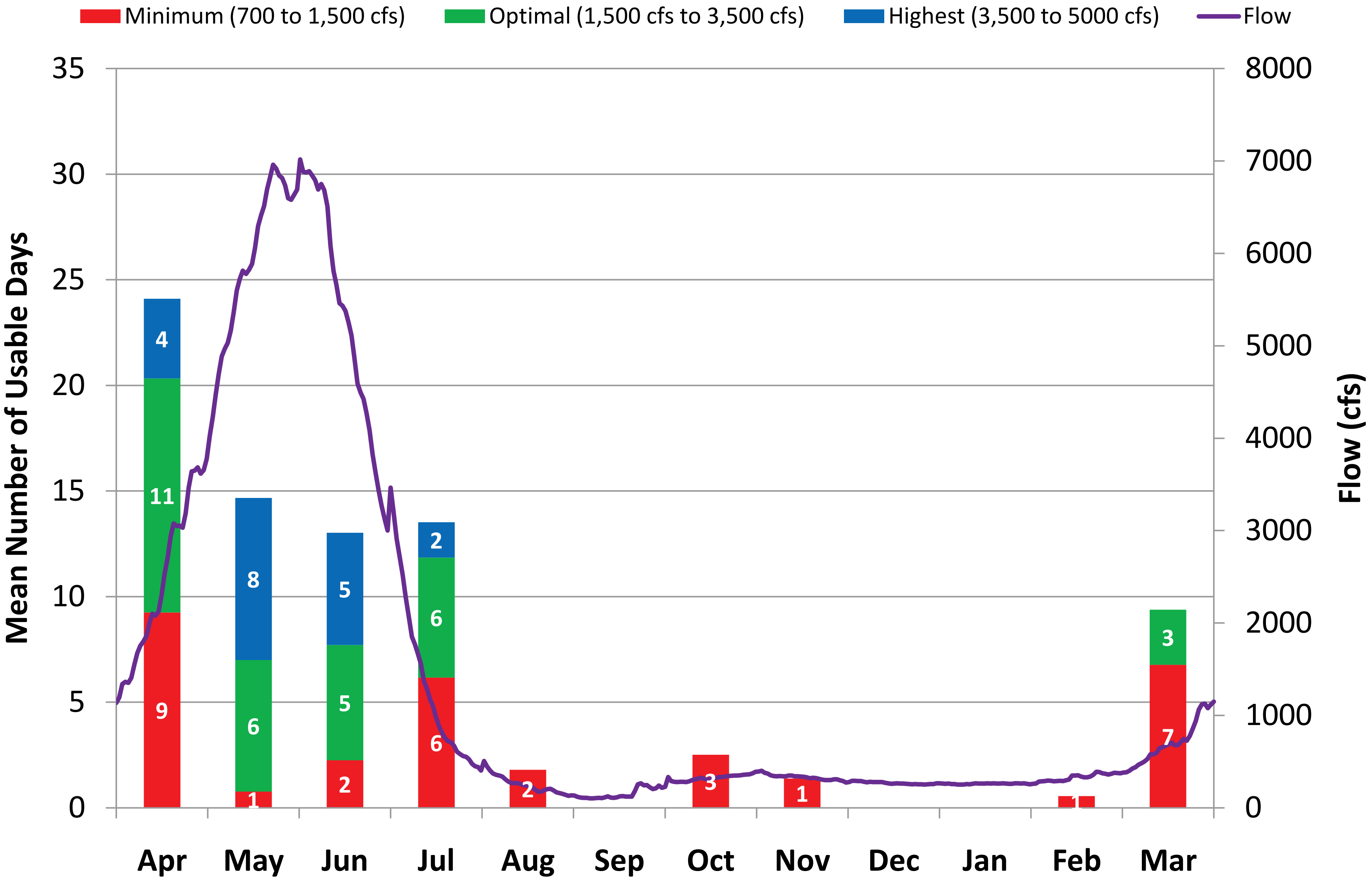


Figure 3-12 Yampa Canyon

Mean Number of Usable Days by Month (1954-2005)

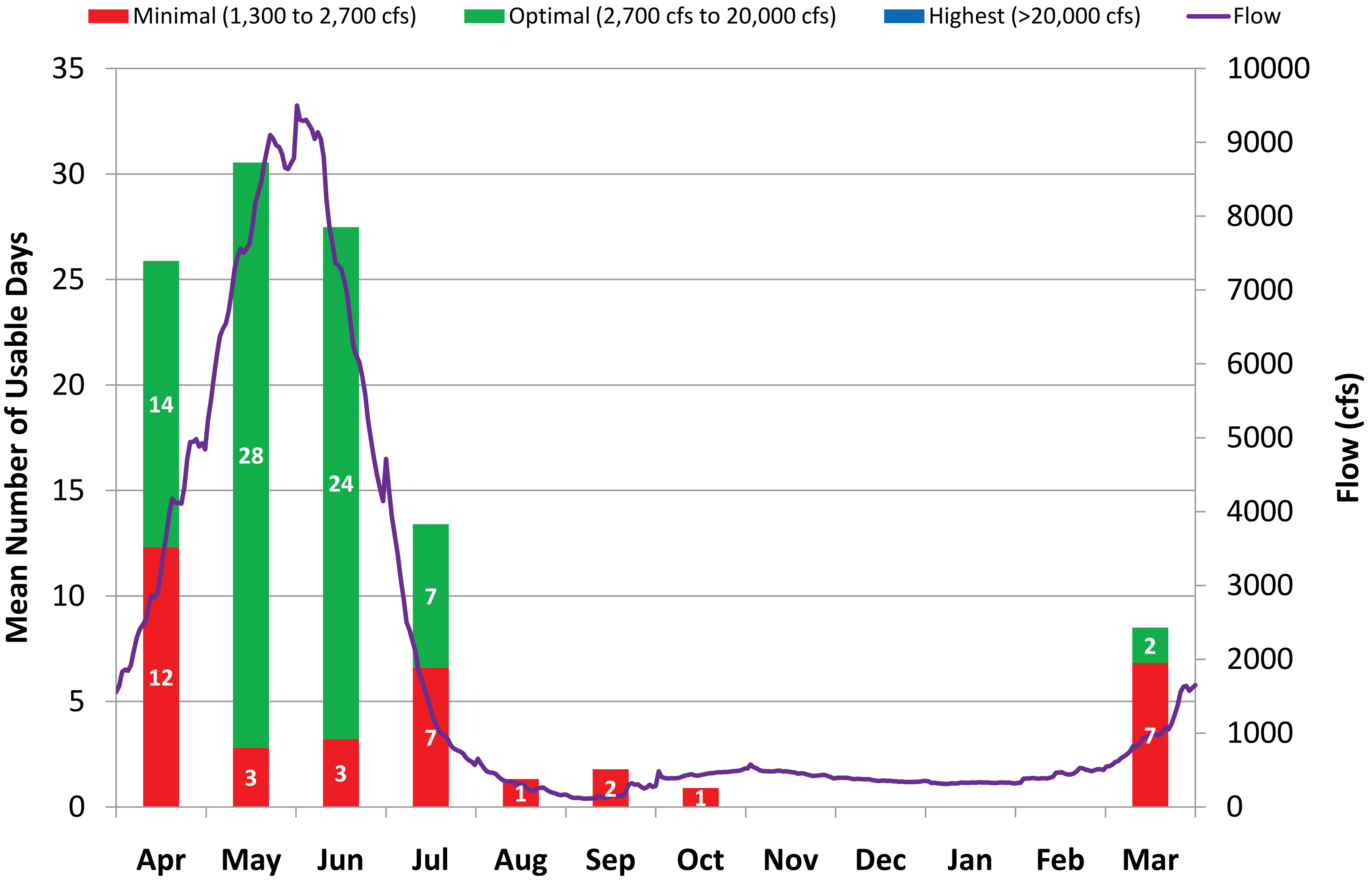


Figure 3-13 Gates of Lodore

Mean Number of Usable Days by Month (1975-2010)

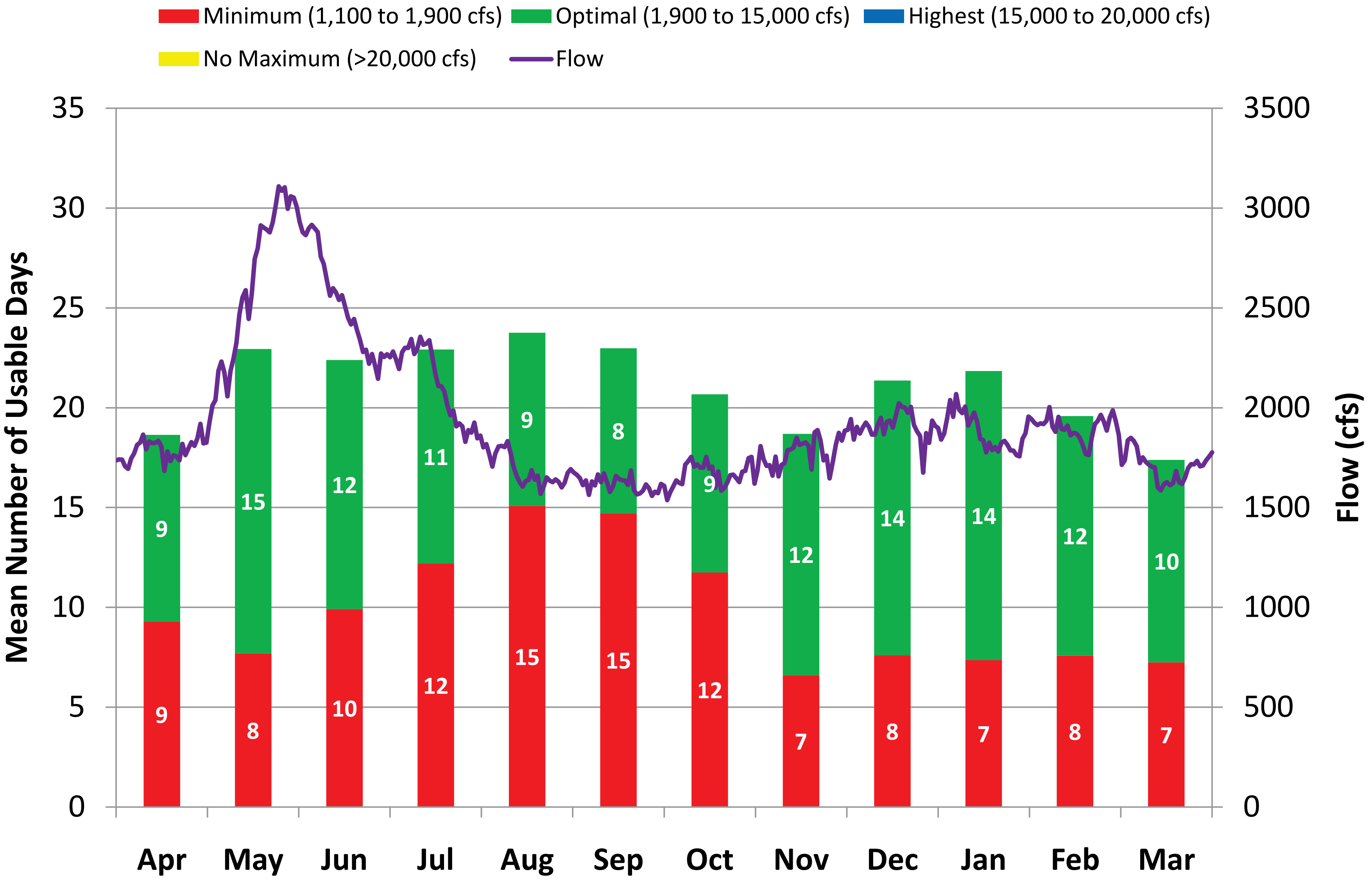
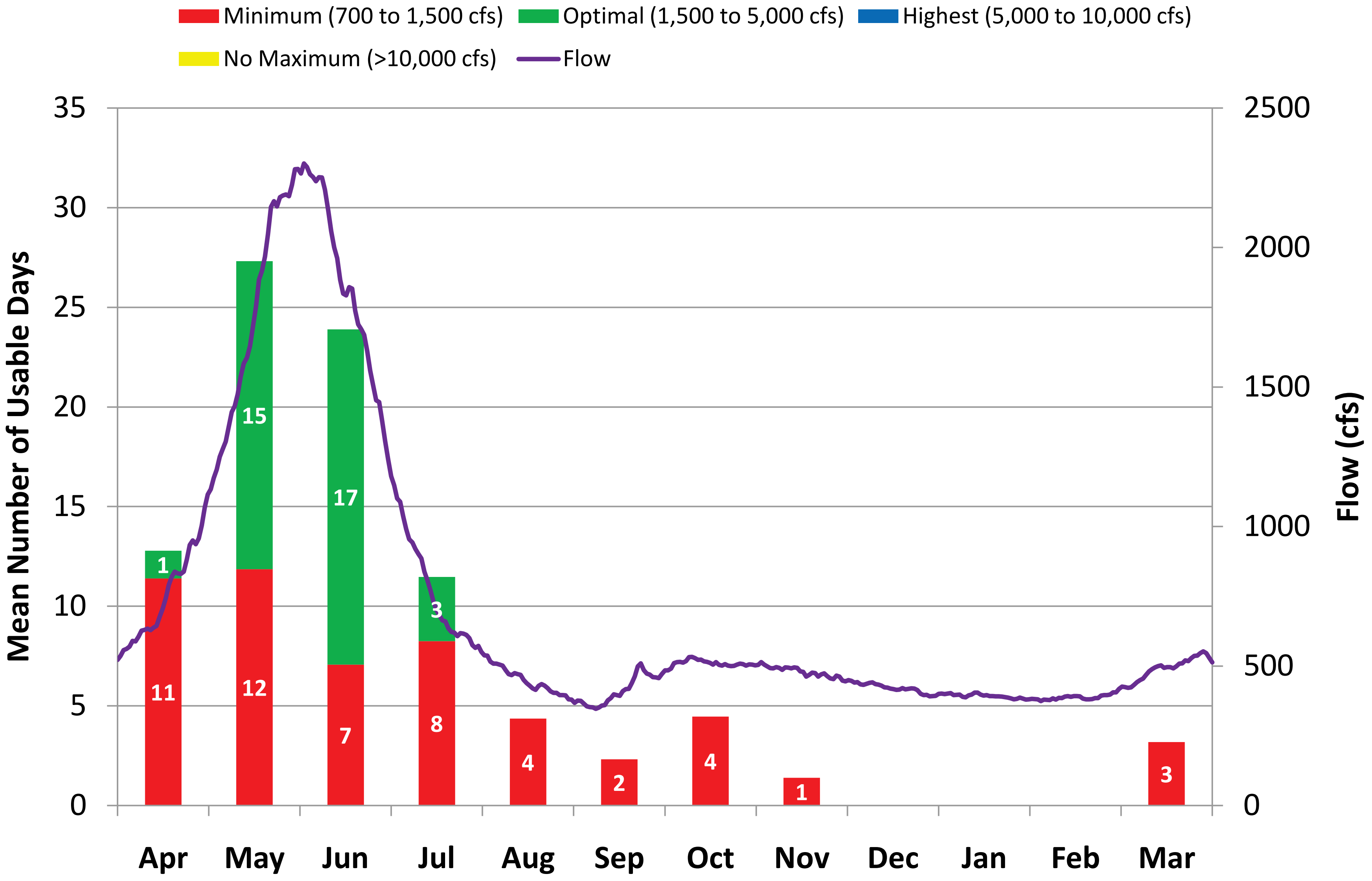


Figure 3-14 White River Rangely to Bonanza
Mean Number of Usable Days by Month (1983-2010)



Section 4

Conclusions and Recommendations

4.1 Capabilities and Limitations of the WFET

Because of the sensitivity to and the potential to misunderstand WFET results, it is worth reiterating a few key points described in Section 2.2 (*Applications, Capabilities, and Limitations of the WFET*):

- Due to the complexity of determinant factors and ecological response, the WFET does not predict the structure and function of an ecological community under past or future conditions. In other words, when a high or very-high flow-ecology risk is indicated by the WFET, there is a higher likelihood of a risk of change to the attribute being considered due to flow management, but the WFET does not indicate that a change or problem necessarily exists.

- The WFET does not speak to the value of a given change in a resource. For example, it does not address whether or not a change in cottonwood establishment is desirable or not.

- The WFET has been developed to identify the risk of ecological change due to flow management, but is insufficient to quantify nonconsumptive water needs on a site-specific basis.

- The WFET is only one tool in the toolkit for assessing environmental condition as it relates to flow management. By design, the WFET is best employed for coarse-level, basinwide assessments, such as screening for potential problem areas or planning for future scenarios. For detailed understand of ecological conditions at a specific location, other tools are recommended.

4.2 Conclusions

Following are the conclusions for the Yampa-White Basin WFET Study based on the approach and results presented in Sections 2 and 3 of this report:

- Flow-ecology relationships were developed for trout, native warm-water fish, and cottonwood (riparian) attributes. Efforts to validate these relationships using data from other studies has supported these flow-ecology relationships as generalized models that can be used in the future to provide a watershed scale understanding of water management as it relates to support of nonconsumptive needs.

- Flow-related risk to trout, native warm-water fish, and riparian attributes was examined under current water management conditions and for future conditions (Yampa River only) following depletions described in the PBO (USFWS 2005). The watershed scale, science-based maps of flow-related ecological risks throughout the drainage correspond well with current understanding of impacts resulting from flow management.

- In general across the White River Basin, trout, warm-water fish, and riparian attributes in the White River Basin have low to moderate flow-ecology risk. Therefore, for the White River Basin, it can be assumed that current river management and water rights are supportive of nonconsumptive needs during both low flow and high flow conditions.

- Most locations across the Yampa River Basin have trout, warm-water fish, and riparian attributes with low to moderate flow-ecology risk. However, there are several locations where high flow-ecology risk is indicated and a couple of locations where very high flow-ecology risk is indicated. The one consistent pattern that emerges from these higher risk locations is that native warm-water fish in the lower part of the river are at-risk due to late season low flows. This finding is consistent with findings and actions in the Upper Colorado River Basin Endangered Fish Recovery Program, which has augmented baseflows for the endangered fish through the enlargement and management of Elkhead Reservoir. Thus, for the Yampa River Basin, it can be assumed that current river and water rights are generally supportive of nonconsumptive needs, but there is one known cause for concern in the lower basin and a few locations higher in the basin where risk may be indicated.

- Natural information was developed for whitewater boating attributes. Whitewater recreation information was summarized for 16 river segments in the basin. A usable days analysis was completed for each of the segments and these results can be utilized in the future to understand how the amount of usable days may vary in the future due to changes in water management.

4.3 Recommendations

Following are recommendations based on study results:

- In the near term, use the WFET in conjunction with the focus area map and the process described above to identify strategies and implementation plans for long-term protections. For example, WFET can be used to identify opportunities under the Alternative Transfer Methods project that has launched in the Yampa and White River Basins.

- In the medium and long term, use the WFET and recreational flow analysis results to analyze scale and distribution of expected flow-related risk to nonconsumptive attributes resulting from new development projects, a Compact call, and/or climate change. For example, in developing a long-range water security plan, the WFET in combination with other tools and data can be used to minimize impacts to nonconsumptive resources, and potentially can be used to identify opportunities for restoration of flow conditions needed to support nonconsumptive attributes.

- Bear in mind that WFET and recreational analysis conducted during the study do not address every issue affecting nonconsumptive outcomes. Flow-related decision-making should be embedded in a framework of planning for all factors affecting these outcomes.

Section 5

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Appendix A

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

Freshwater Biology

Freshwater Biology (2010) **55,** 147–170 doi:10.1111/j.1365-2427.2009.02204.x

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 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

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scientists, water-resource managers and stakeholders to analyse and synthesise 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 user-defined region to develop scientifically defensible and empirically testable 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 modelling to build a 'hydrologic foundation' of baseline and current hydrographs for stream and river segments throughout the region. Second, using a set of ecologically relevant flow variables, river segments within the region 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 hydraulic 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, expert knowledge and field studies across gradients of hydrologic alteration.

4. Scientific uncertainty will exist in the flow alteration–ecological response relationships, 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 can be used both to guide basic research in hydroecology and to further implementation of more comprehensive environmental flow management of freshwater sustainability on a global scale.

Keywords: environmental flows, hydroecology, hydrologic modelling, river management, streamflow classification

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 fibre as well as places for recreation, tourism and other cultural activities (Postel & Carpenter, 1997; Naiman *et al.*, 2002; Dyson, Bergkamp & Scanlon, 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 optimise 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 recognised as a major social-scientific challenge of the 21st century (Palmer *et al.*, 2004).

Environmental flows are defined in the Brisbane Declaration (http://www.riverfoundation.org.au/images/stories.pdfs/bnedeclaration.pdf) as the 'quantity, timing and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihood and well-being that depend on these ecosystems'. 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; Poff, 2010), and this understanding has contributed to the implementation of environmental flow management on thousands of river kilometres worldwide (Postel & Richter, 2003). Despite this tangible progress, millions of kilometres 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 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 recognised at national and international scales (Petts, 1996; Dyson *et al.*, 2003; GWSP, 2005; NSTC, 2004; CAWMA, 2007; Brisbane Declaration, http://www.riverfoundation.org.au/images/stories.pdfs/bnedeclaration.pdf). 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 (Brown & Joubert, 2003; Tharme, 2003; Annear *et al.*, 2004; Arthington *et al.*, 2004; King & Brown, 2006). Thus, a key challenge in securing freshwater ecosystem sustainability is synthesising 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 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 analyse and synthesise 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, 2010). Thus, both ecological theory and abundant evidence of ecological degradation in flow-altered rivers support the need for environmental flow management. Certainly, environmental factors other than streamflow (including temperature, water quality, sediment and invasive species) also regulate riverine ecosystem structure and function, as has been well recognised (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 decline further (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, Henriksen & Nieswand, 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 and Wood, 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 modelled 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) and hopefully sustained.

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 at: http://conserveonline.org/workspaces/eloha.

The scientific process in the ELOHA framework

The ELOHA framework involves a number of interconnected steps, feedback loops and iterations (Fig. 1). Relationships between flow alteration and ecological characteristics for different river types constitute 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, recognising 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 our express intent 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 support decisions at broad regional scales (Acreman & Dunbar, 2004; Booker & Acreman, 2007). 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.

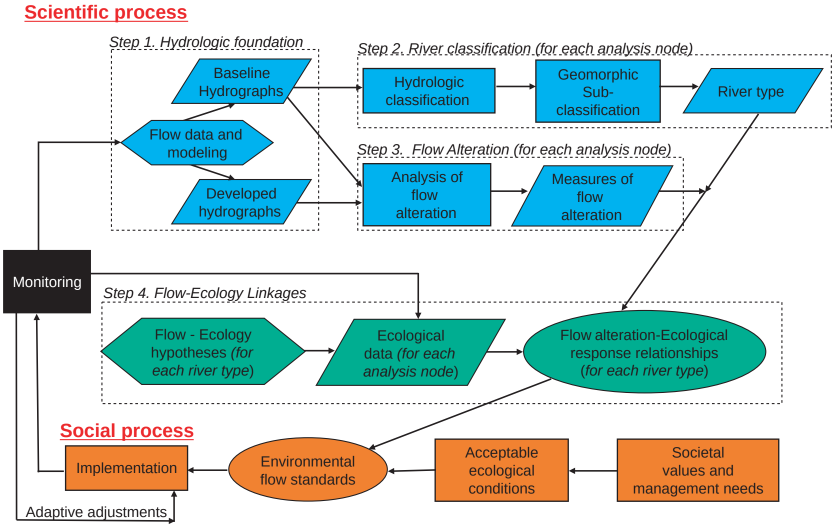

Fig. 1 The ELOHA framework comprises both a scientific and social process. Hydrologic analysis and classification (blue) are developed in parallel with flow alteration–ecological response relationships (green), which provide scientific input into a social process (orange) that balances this information with societal values and goals to set environmental flow standards. This paper describes the hydrologic and ecological processes in detail, and outlines the scientist's role in the social process.

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 modelling 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 characterisations of management interest.

The coupled baseline and developed hydrologic time-series constituting 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 serve as independent variables in developing flow alteration–ecological response relationships (described in Formulating flow alteration–ecological response relationships for environmental flows 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 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. 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.

Figure 2 illustrates the general approach for building the regional hydrologic foundation. Briefly, the approach relies on region-specific combinations of streamflow gauge analysis and hydrologic modelling. Existing streamflow gauge records for a selected time period are segregated into those that represent baseline conditions and those that represent developed conditions. Differences between baseline and developed conditions are characterised in terms of

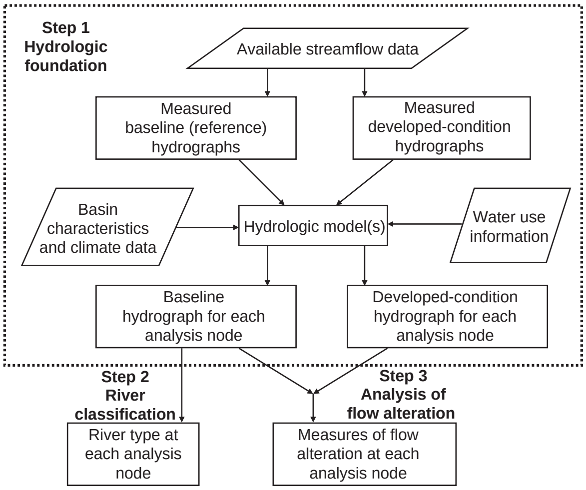

Fig. 2 Steps for developing the hydrologic foundation (ELOHA step 1 inside dashed box), showing how the resulting hydrographs are used to classify river types (ELOHA step 2) and calculate flow alteration (ELOHA step 3) at each analysis node.

statistical departures in the ecologically relevant components of the two flow regimes. At ungauged analysis nodes and for time periods not represented in the period of record, statistical techniques (Sanborn & Bledsoe, 2006; Stuckey, 2006; Zhang *et al.*, 2008; Carlisle *et al.*, 2009) can be used to estimate flow metrics, or hydrologic simulation models of rainfall–runoff and other catchment processes (Singh & Woolhiser, 2002; Wagener, Wheater & Gupta, 2004; Blöschl, 2005; Kennen *et al.*, 2008) can be developed to generate flow time-series from which metrics can be extracted. In heavily modified catchments, simulation models can be especially useful in estimating baseline flow regimes through removal of flow extractions and reservoirs (e.g. Yates *et al.*, 2009), as well as adjusting various model parameters (e.g. infiltration, interception, routing) to represent past land cover conditions (Beighley, Melack & Dunne, 2003). For rapidly changing land uses (e.g. urbanisation), developed-condition hydrographs could be modelled 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 modelled, 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 & Slack, 1999; McCabe & Wolock, 2002), it is desirable to adopt a single time period (e.g. 10–20 years) as a climatic reference period for which baseline and developed-condition streamflows are synthesised and modelled. 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* estimation of model parameters 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). An alternative to regionalisation of model parameters to simulate streamflow time series at ungauged locations is regionalisation of streamflow characteristics to generate flow statistics, which allows for explicit estimation of uncertainty (see Zhang *et al.*, 2008). 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 *et al.* (2009) 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.

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 features. 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 (Lytle & Poff, 2004; Arthington *et al.*, 2006), 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 modelling 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 building a hydrologic foundation. 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 (http://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. small within-type variability) and interpretability (i.e. differences among types). 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. Most previous regional to continental hydrologic classifications have used four to 12 classes, depending on geographic extent, climatic and geologic variation or inclusion of other environmental factors (e.g. Poff & Ward, 1989; Poff, 1996; Snelder & Biggs, 2002; Kennen *et al.*, 2007, 2009; Acreman *et al.*, 2008; Kennard *et al.*, 2010).

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 (Richter *et al.*, 1996; Poff *et al.*, 1997; 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 catchment 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 & Poff (2003) contained in the HAT 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' 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, although 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 physical 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 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 physical characteristics, such as constrained versus alluvial channels or sand-bedded versus 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; Vaughan *et al*., 2009).

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 modelling hydrographs at ungauged locations, for both baseline and current conditions. Promising approaches (i.e. that are technically feasible and cost-effective) include catchment 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*., 2009). Another approach, by Kennen *et al.* (2008), couples runoff modelling 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 catchment 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 catchment management, refer to Singh & Woolhiser (2002).

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 (see Arthington *et al*., 2006). These relationships are hypothesised 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 formalised, 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.*, 2010).

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 hydroecology is comprised of both comparative and experimental studies that relate ecological processes or aspects of ecosystem structure and function to one or more hydrologic variables (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, 2010). 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 (e.g. Bunn & Arthington, 2002).

A guiding principle for such model development from the existing hydroecological 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, 2010). 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 urbanisation) favour fish species with more generalised versus specialised 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 of non-native fish (Fausch *et al.*, 2001), whereas the loss of such seasonal flooding can promote success of non-native fish species (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 responses begins by posing a series of plausible hypotheses that are based on expert knowledge and understanding of the hydroecological 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. 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, Thoms & Quinn, 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 data sets and in designing monitoring programs or research projects for validating and refining the relationships.

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

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 riparian 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 species with preferences for slow-flowing, shallow-water habitats, whereas fluvial specialists 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
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-colonisation ability
Hyp: Increased flood frequency (in channels) will reduce abundance of young-of-the-year fish, but decline in flood frequency would favour flood-intolerant species
Hyp: A decrease in inter-annual variation in flood frequency (i.e. stabilised 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 stream channel 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

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 & Richter (2007).

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 categorised in a variety of ways: taxonomic identity, level of biological organisation (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 transient or 'equilibrial' conditions). 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). Indicators of social value may also be used to assess flow alteration. The response times of these multiple possible response variables 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

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

Mode of response
- Direct response to flow, e.g. spawning or migration
- Indirect response to flow, e.g. habitat-mediated

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)

Rate of response
- Fast versus slow
  - Fast: appropriate for small, rapidly reproducing, or highly mobile organisms
  - Slow: long-life span
- Transient versus equilibrial
  - Transient: establishment of tree seedlings, return of long-lived adult fish to potential spawning habitat
  - Equilibrial: reflect and end-point of 'recovery' to some 'equilibrium' state

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, behavioural, life-history adaptations (e.g. short-lived versus long-lived, reproductive guilds)
- Habitat requirements and guilds
- Functional diversity and complementarity

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

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

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 by the ability to monitor these various responses over time.

Ideal ecological (including habitat) response variables are (i) sensitive to existing or proposed flow alterations; (ii) amenable to validation with monitoring data and (iii) 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 for fish (e.g. Fausch, Karr & Yant, 1984; Kennard *et al.*, 2006a,b) or benthic invertebrates (e.g. DeGasperi *et al.*, 2009), and the lotic-invertebrate index for flow evaluation 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 utilisation 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, Bovee & Waddle, 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 method (e.g. downstream response to imposed flow transformation: Arthington *et al.*, 2003; King *et al.*, 2003).

In general, developing characterisations 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. 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 generalise hydraulic habitat relations for specific geomorphic subclasses. For example, Lamouroux (1998), Lamouroux, Souchon & Herouin (1995) and Booker & Acreman (2007) have developed generalised models for depth and velocity at the stream reach scale, and Saraevan & Hardy (2009) present 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 flow alteration can take many forms, as noted by Arthington *et al.* (2006). Based on current hydroecological understanding, we expect the form of the relationship to vary depending on the selected ecological response variable(s), the specific flow metric(s) and the degree of alteration for a given river type. These relationships could follow a number of functional forms, from monotonic to unimodal to polynomial. Different ecological response variables may increase or decrease with flow alteration, and the functional form of the response may depend on whether flow alteration of a particular flow variable increases or decreases. We illustrate how various ecological responses may vary with specific components of flow alteration in Fig. 3, which presents plausible relationships for three river types (Fig. 4). For each river type the reference condition is represented by the range of natural variation for both the flow variable and the ecological variable of

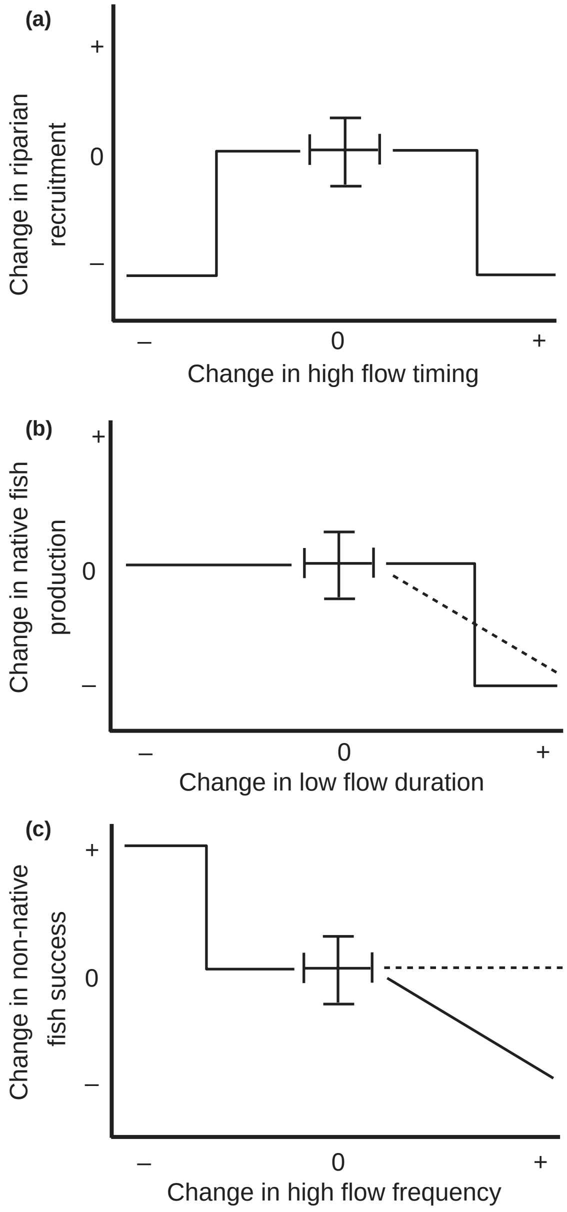

Fig. 3 Illustrative flow alteration–ecological response relationships for each of three river types: (a) snowmelt, (b) groundwater-fed and (c) flashy. For each relationship the change in the flow metric (*X*-axis) ranges from negative to positive with no change representing the reference condition. The response of the ecological variable (*Y*-axis) to the flow alteration across a number of altered sites ranges from low to high. The bracketed space in the centre of the graph represents the natural range of variation in the flow variable and ecological variable in the reference sites. Ecological responses depicted can range in functional form from no change to linear to threshold, depending on the underlying hydroecological mechanisms and, in some cases, on the specific geomorphic context (indicated by dashed line). See text for further explanation and discussion.

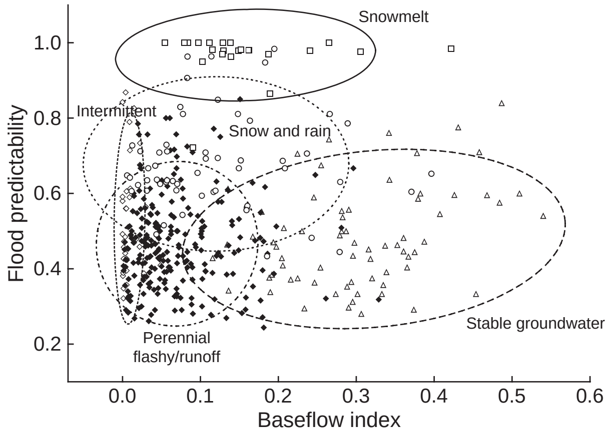

Fig. 4 Plot of five river types in U.S. (modified Olden & Poff, 2003). River types (based on 420 stream gauges) are defined in terms of 11 flow metrics but plotted here in two-dimensional space defined by two of the classification flow metrics (flood predictability and baseflow index). Ellipses reflect 90% confidence intervals and show natural range of variability for the two flow metrics for each of five river types: snowmelt (open squares), snow and rain (open circles), stable groundwater (open triangles), perennial flashy/runoff (closed diamonds) and intermittent (open diamonds – combined harsh intermittent, intermittent flashy and intermittent runoff classes from Poff 1996).

interest, and the ecological response is depicted in terms of deviation from the reference flow condition.

For snowmelt river types (Fig. 4), the successful recruitment of native riparian trees often depends on seed release being coincident with the timing of flows of sufficient magnitude to raft seeds onto suitable riverbank habitat (e.g. cottonwood in the western North America; Mahoney & Rood, 1998). Some alteration of high flow timing can occur and still coincide with seed release; however, if high flows come earlier than seed release (negative change) or if they are delayed until after seed release (positive change) then recruitment is expected to drop off precipitously in a threshold-type response (Fig. 3a).

In stable groundwater-fed streams, low flows generally have relatively short duration (Poff, 1996). Reducing the duration of low flows in these systems would not be expected to have a large effect on native fish (solid line with no slope in Fig. 3b) because low flow stress is generally transient under natural conditions. By contrast, increasing the duration of low flows could dewater habitat and damage native species (see Moyle *et al.*, 2003), perhaps via a threshold-type reduction (solid step-function line in Fig. 3b). However, the effect could depend on geomorphic context. For example, a river with deep pools would offer refuges for fish during extended low flow periods and thus a more gradual and continuous (linear) ecological response would be expected (dashed line in Fig. 3b).

Third, naturally flashy streams and rivers are typified by high frequency or rapid onset of high flows. Non-native species of fish may fail to establish in such streams if they lack behavioural adaptations to rapid onset of erosive flows (Meffe, 1984) or if the vulnerable juvenile life stage is present during periods of peak flows (Fausch *et al.*, 2001). Figure 3c shows how a reduction in high flow frequency could benefit non-native fish species, possibly as a threshold response by allowing a sufficient number of juveniles to escape mortality and establish large populations. By contrast, increasing high flow frequency would be expected to depress the success of poorly adapted fishes (solid line with negative slope); however, high structural habitat heterogeneity or the presence of within-channel refuges (pools, backwaters) could provide hydraulic refuges and ameliorate this response (dashed line).

These examples illustrate the process of linking particular ecological responses to specific types of flow alterations in the context of natural flow variability for different river types. 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, which would also represent testable hypotheses of response to hydrologic alteration. Certainly a large number of possible flow alteration–ecological response relationships can be postulated and supported from the scientific literature. For any particular application of ELOHA these will reflect the diversity of river types and ecological response variables of interest in a given region.

One important reason for developing a flow regime classification is that the form and direction of an ecological response to flow alteration is hypothesised 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 (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 two-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 stabilisation 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 enable, 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 versus 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 (see Poff & Zimmerman, 2010). 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. The benchmarking approach of 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 judgement and includes multiple ecological indicators, as is commonly employed in performing river-specific environmental flow assessments based on expert judgement in South Africa (Brown & Joubert, 2003; Tharme, 2003), Australia (Cottingham *et al.*, 2002; Arthington *et al.*, 2004) 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 can support 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 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 response(s) in terms of some multivariate hydrologic metric(s) that describes overall flow alteration (e.g. using principal components analysis as in Black *et al.*, 2005). 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 prioritise for management. In the face of multiple possible management targets, 'paralysis' can be avoided by keeping in mind the motivating objectives of the selection process for hydrologic variables. Flow metrics ideally 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 reasonable confidence, that best define the hydrologic and ecological 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 depicts the major facets of the flow regime and explains much of the observed variation in ecological response to particular kinds of flow alteration in each river flow type.

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, 2008); 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 minimise 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 in many cases. This could be done either by further stratification (e.g. based on water temperature or water quality; see Olden & Naiman, 2010) 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.

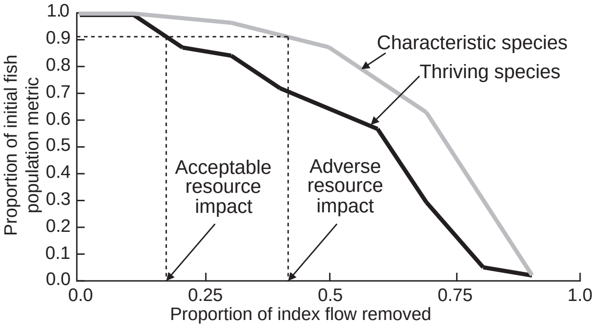

Fig. 5 Progression from flow alteration–ecological response relationships to environmental flow standards (modified from MGCAC, 2007). Using existing fish population data across a gradient of hydrologic alteration, scientists developed two flow–ecology relationships between populations of 'thriving' and 'characteristic' fish species versus proportion of 'index' flow (median August discharge divided by mean annual discharge) flow reduction in 11 stream types in Michigan, U.S.A. A diverse stakeholder committee then proposed a 10% decline in the thriving fish population index as an acceptable resource impact, and a 10% 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.

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. For example, 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 recognise 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 synthesises all of the controls – both natural and engineered – on streamflow patterns into one usable database. Thus, it can be useful not only for establishing flow–ecology relationships, but also for integrating them into the social decision-making process. In principle, scientists and managers could use the hydrologic model to test various stakeholder-developed scenarios for coordinating and optimising all geographically referenced water uses in a basin, while maintaining environmental flows. The model should also be able to incorporate predicted hydrologic impacts of climate change. By accounting for the cumulative effects of all water uses, the model could be used to assess the practical limitations to, and opportunities for, implementing environmental flow targets at multiple nodes simultaneously. This would support efforts to prioritise development of restoration projects, optimise 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, such a model could 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 alteration-ecological response 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 recognise 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 recognise 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 judgement 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 for freshwater sustainability. Further, through future adaptive learning and research the ELOHA framework can provide a foundation for refining efforts to optimise the tradeoffs inherent between resource exploitation and resource conservation (Dudgeon *et al.*, 2006).

We have emphasised in this paper that scientific knowledge and theory pertaining to flow alteration–ecological response principles has advanced markedly in recent decades, and the calibre of data and 'professional judgement' 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 initial 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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Appendix B

Colorado Decision Support Model Summary for Yampa-White WFET Project

Modified Straight Line Diagram for Modifications to Elkhead and the Yampa Management Plan EFS in the CDSS Yampa StateMod Water Allocation Model

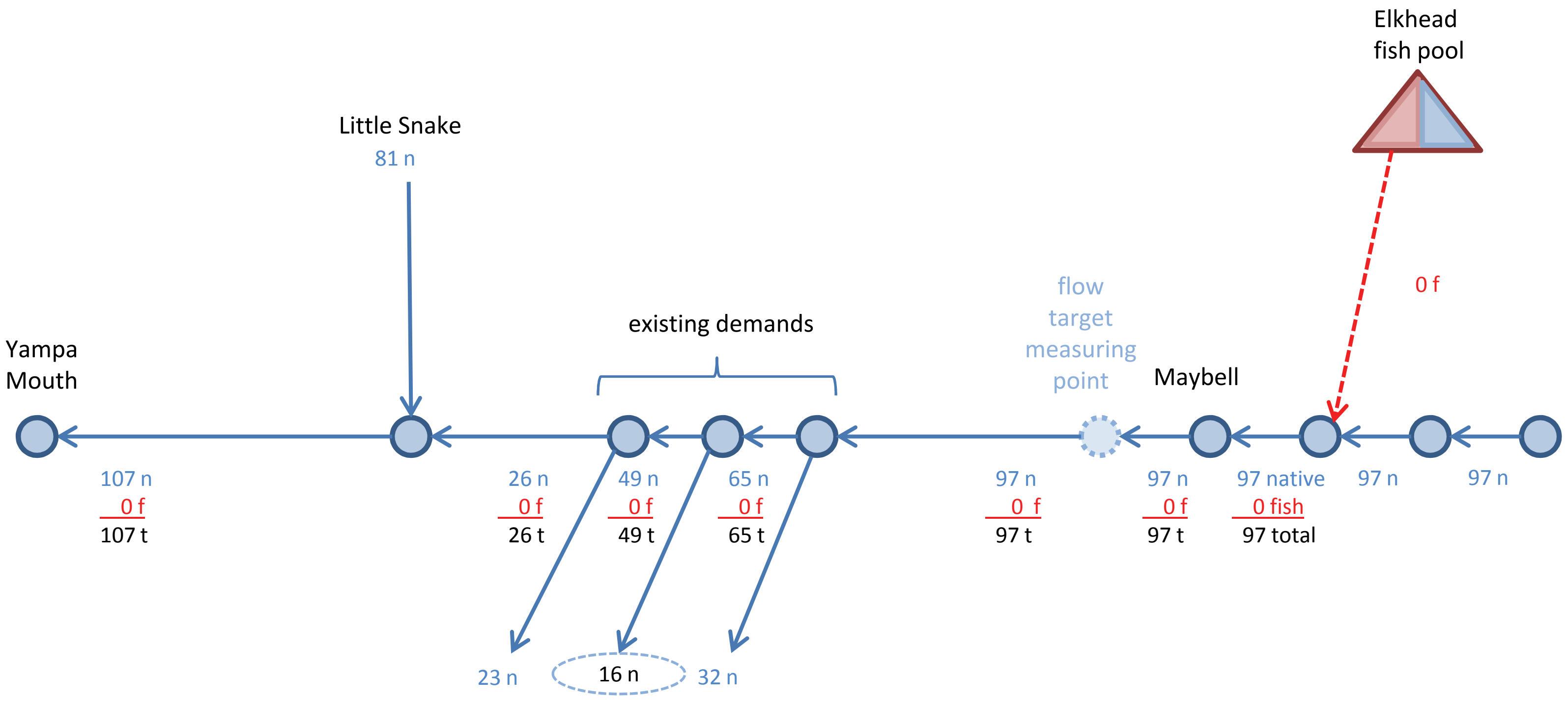
March 30, 2012
Yampa-White Watershed Flow Evaluation Tool
CDM Smith (Nicole Rowan and Mark Hoener)

- Original diagrams and concepts compiled by High Country Hydrology (John Winchester) Ecological Engineering International (Brian Bledsoe) for The Nature Conservancy, Colorado River and State of Colorado Programs on January 14th, 2011.
- Assumption that minimum flows through the ESF reach (also referred to as fish flows) are 49 cfs.
- Critical location is upstream of the Little Snake confluence with the Yampa River.
- This hypothetical situation assumes that a 23 cfs release is made from Elkhead to increase flows through the ESF reach.
- The following slides show the impact of different scenarios on the flow through each reach.

CDSS Modeling Assumptions

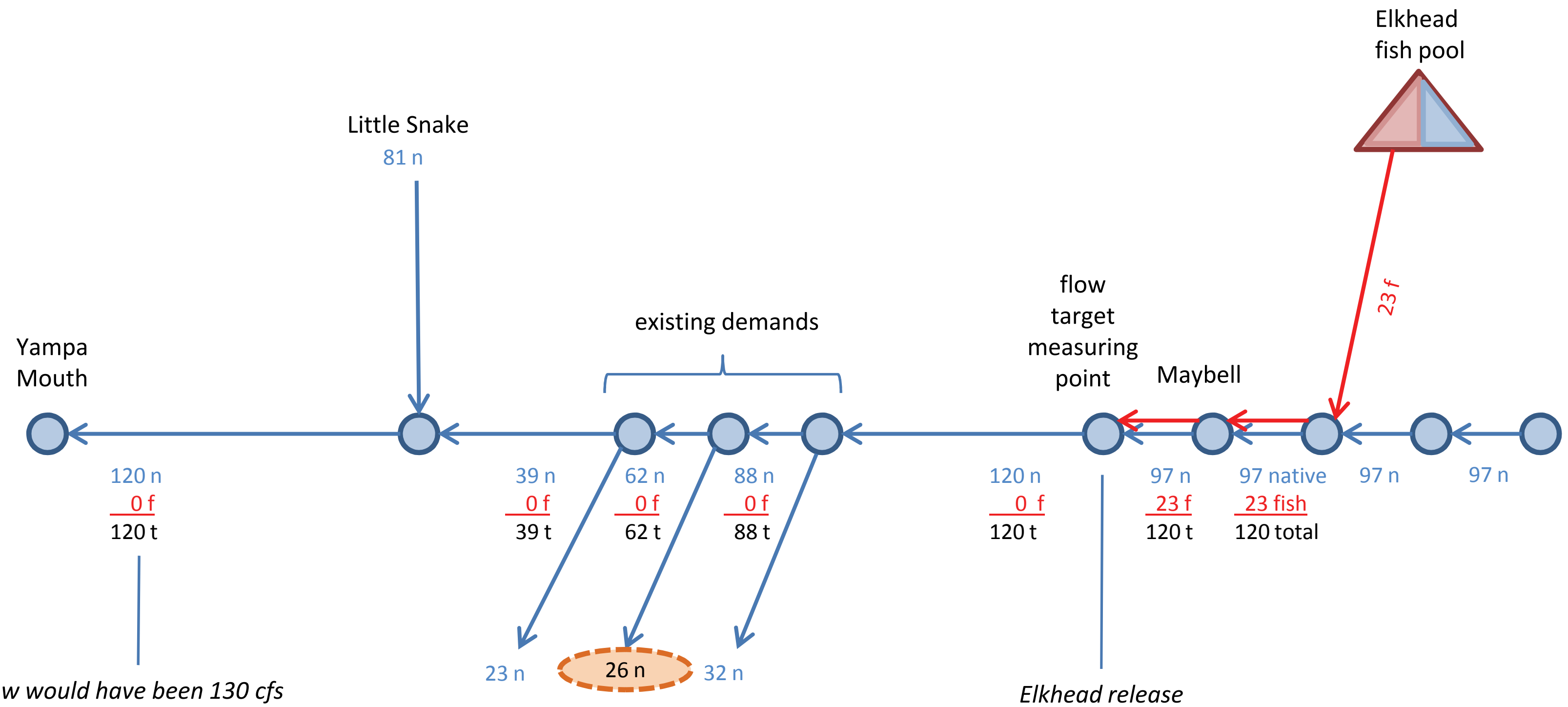
- See following slides for how Elkhead releases are "sheparded" and not diverted
- Modified existing CDSS model (CWCB website version 12.29.30 dated 2/4/2010)
- For Elkhead operations:
  - Not transit losses associated with reservoir releases
  - Reservoir releases limited to 50 cfs
  - Reservoir releases not limited to annual volume (e.g., 7,000 ac-ft)
- CDSS modeling mechanism utilized was "carrier" to shepard flows

StateMod without fish flow releases



StateMod without ISF for Fish Flows

as currently modeled in StateMod



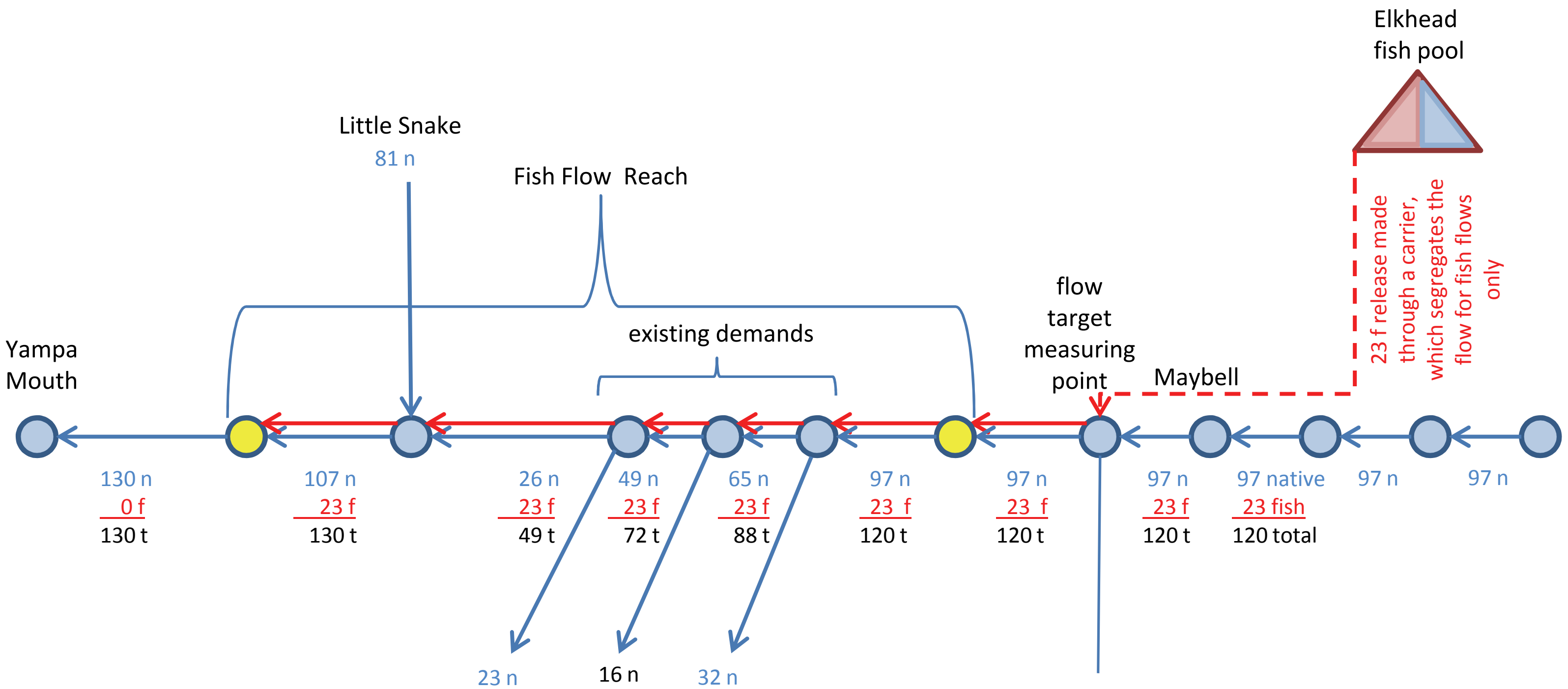
Flow would have been 130 cfs at mouth if 23 cfs had been shepherded past intervening demands

Junior or senior right diverts an additional 10 cfs due to fish flow release

Elkhead release becomes native flow available for diversion below measuring point for flow target

StateMod with In Stream Flow Reach for Fish Flow

Modified StateMod model used for WFET

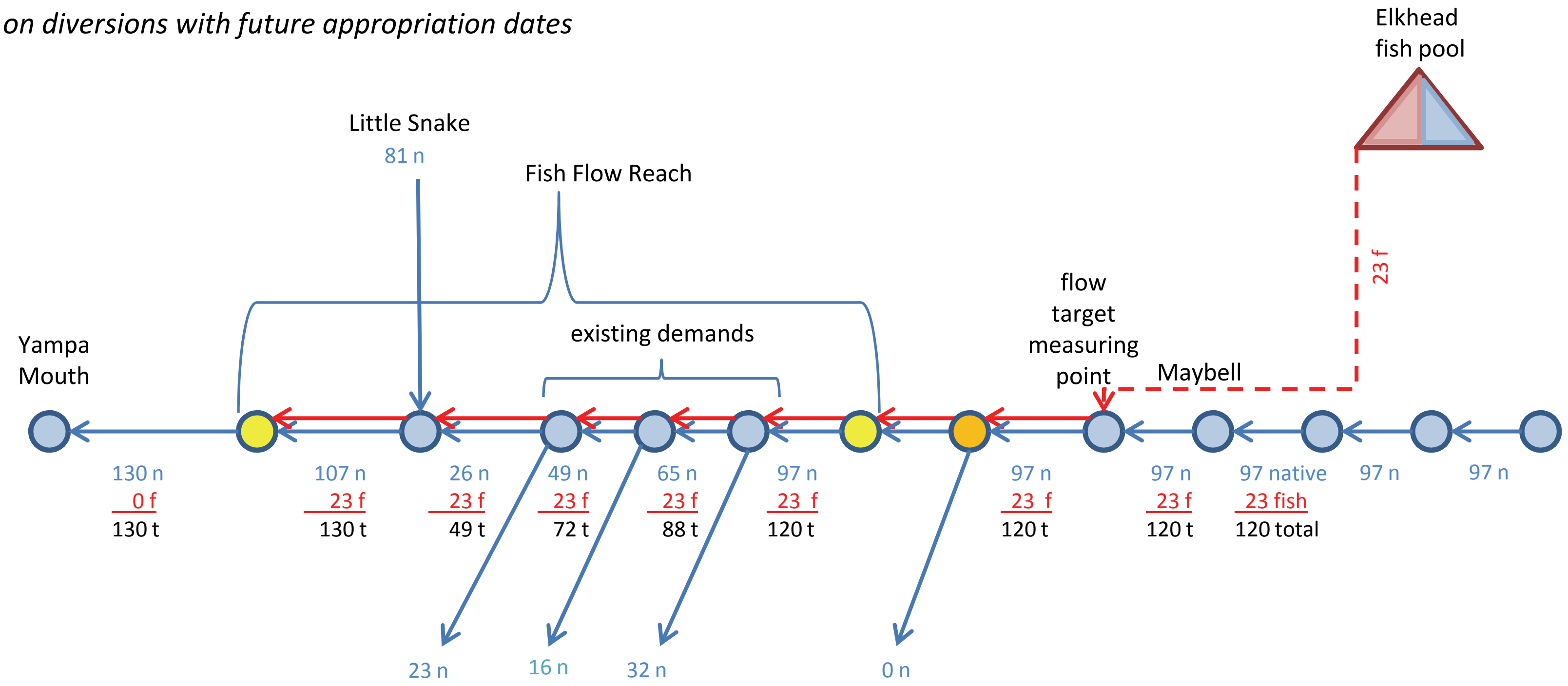


Junior existing right cannot divert the full 26 due to segregated fish flow releases

Release made through a carrier to the flow target measuring point

StateMod with In Stream Flow Reach for Fish Flow and a New Junior Siversion

Effect on diversions with future appropriation dates



Fish flows through the ISF are senior to all future rights. Junior Rights are called out due to the critical reach upstream of the Little Snake Confluence

Appendix C

Geomorphic Subclassification

A Geomorphic Valley Classification for Fluvial Riparian Areas

Brian P. Bledsoe, Ph.D., P.E. and Erick A. Carlson

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Executive Summary

The upper Colorado River basin contains a diverse mosaic of geomorphic settings and fluvial riparian ecosystems. From the steep, v-shaped and glacial valleys of the high country to the gentle gradients and expansive floodplains of lowland alluvial valleys, geomorphic setting mediates the relationship between hydrology and riparian ecosystems. CSU has collaborated with the US Forest Service over the last 4 years in the development of a geomorphic valley classification (GVC) for describing the key geomorphic factors that influence riparian systems across large regions. The classification is geographic information systems (GIS) based and delineates different geomorphic valley settings using energy, hillslope coupling, and lateral confinement as the primary diagnostic characteristics. The GVC derives its class descriptions from geomorphic thresholds corresponding to significant transitions in the physical processes and boundary conditions that give rise to distinct floodplain and channel forms, disturbance regimes, and ecological attributes (Table ES1).

Table ES1: Valley classification names and attributes of the GVC.

Valley Class Name	Energy / Valley Gradient	Valley Bottom Width / Coupling / Confinement	Hillslope Gradient	Energy Potential
Headwaters	> 4%	$< (2\,L_D + W_{BF})$	Both > 30%	High
High-energy Coupled	> 4%	$< (2\,L_D + W_{BF})$ or $< (L_D + W_{BF})$	Both or at least one > 30%	High
High-energy Open	> 4%	$> (2\,L_D + W_{BF})$	Both or at least one > 30%	High
Moderate-energy Confined	0.1-4%	$< 7\,W_{BF}$	Variable	Moderate

Moderate-energy Unconfined	0.1-4%	> 7 W_{BF}	Variable	Moderate
Canyon	Variable	> 3 W_{BF}	> 70%	Moderate to High
Gorge	Variable	< 3 W_{BF}	> 70%	Moderate to High
Glacial Trough**	< 4%	> (2 L_D + W_{BF})	~ 10-% initially steepening to > 30%	Moderate to Low
Low-energy Floodplain	< 0.1%	> 7 W_{BF}	Generally < 30%	Low

L_D – length of debris runout W_{BF} - width of channel at bankfull stage

** Defined as valleys with the given characteristics, lying above the elevation of the most recent glacial activity

In the GVC, energy refers to the hydraulic power available to scour and shape valley bottoms and the channels they contain. Energy is characterized using unit stream power or valley slope as its surrogate. The slope thresholds selected for distinguishing between valley energy types correspond to widely recognized shifts in hydro-geomorphic processes. For example, valleys steeper than 3-4% slope tend to contain confined step-pool and cascade channels with varying degrees of hillslope coupling. As valley slopes become less than 3-4%, the channel types gradually shift to broader floodplains containing plane bed, pool-riffle, and sandy streams.

Coupling refers to the proximity of the hillslopes to the channel and the likelihood that landslides and debris flows on those slopes may move directly across the valley bottom into the stream channel at the slope base. In coupled settings, the channels and the riparian communities occurring along them may be more influenced by materials transported directly from hillslopes (colluvium) than by materials transported from upstream by water (alluvium). In uncoupled settings, sediment recruitment and transport largely become consequences of erosion of the streambed and banks.

Finally, confinement refers constraints on the planform (e.g. meandering, braiding) and lateral adjustability of stream channels. It is quantified by comparing the width of the valley bottom available for channel meandering and migration versus the size of the channel. A sinuous channel typically requires a minimum valley bottom width of approximately seven channel widths to freely meander. By distinguishing between coupling and confinement, the GVC provides a tool for mapping locations where hillslope processes may largely control riparian attributes versus those locations where fluvial processes dominate, as well as a method for distinguishing the degree to which valley bottom widths constrain channel patterns and floodplain processes.

A Geomorphic Valley Classification for Fluvial Riparian Areas

Brian P. Bledsoe, Ph.D., P.E. and Erick A. Carlson

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Background

Riparian zones border flowing and permanent water bodies and are of great ecological and economic importance. These biologically-rich and geomorphically-dynamic areas perform numerous ecological functions that critically influence species distributions, water quality, hydrologic processes, and biogeochemical cycling. The upper Colorado River basin contains a diverse mosaic of geomorphic settings and fluvial riparian ecosystems. From the steep, v-shaped and glacial valleys of the high country to the gentle gradients and expansive floodplains of lowland alluvial valleys, geomorphic setting mediates the relationship between hydrology and riparian ecosystems.

For over a decade, riparian scientists and managers have identified a need for a robust, broadly-applicable fluvial classification for stratifying riparia across large regions (NRC 2002). In 2006, US Forest Service and Colorado State University began a project that aims to synthesize the large body of information on fluvial and riparian systems into a novel, process-based classification fluvial riparian zones in the western US. The specific five goals of this project were to: 1) examine existing fluvial classifications to identify gaps, opportunities for integration, and potential improvements to aid management of fluvial riparian systems; 2) synthesize previous knowledge in developing an *a priori* classification that is process based, hierarchical, and geographic information systems (GIS) based; 3) develop three suites of procedures focused on quantifying system energy, hillslope coupling, and lateral confinement; and 4) test the functionality and accuracy of the classification in several USFS units.

This document summarizes the resulting Geomorphic Valley Classification (GVC) that focuses on energy, hillslope coupling, and lateral confinement as primary diagnostic characteristics. The GVC framework provides a widely-transferable framework for stratifying fluvial systems in the context of management, planning, and monitoring. For example, the GVC can aid in identifying hydro-geomorphically similar reference locations for monitoring, mapping of critical resources for future inventorying activities, and identifying resources at risk from human influences.

In this study, three sets of GIS procedures were developed for stratifying fluvial riparian settings in the upper Colorado River Basin. These procedures measure the key geomorphic descriptors in the GVC using widely-available 10-m digital elevation models (DEMs). The GVC derives its class descriptions from process thresholds that identify significant adjustments to the flow of energy and matter in systems that result in unique landforms, disturbance regimes, and ecological attributes. The open framework of the GVC prompts the user to adjust specific values for hillslope stability, colluvial debris run-out, and fluvial network density. Field testing of the GVC was completed at 42 sites in five ecoregions across the western United States to assess the correspondence between classifications completed using GIS and field data as described below.

Conceptual and Theoretical Framework

The GVC identifies thresholds that describe distinctions between dominant fluvial processes and groups variability into functionally unique classes. The defining attributes of each class are related to the geomorphic processes most influential at a site with a particular set of physical characteristics. The characteristics are related to processes that directly result in the forms observed, disturbances likely to occur, and other physical constraints on the biota inhabiting the site. A connection between the geomorphic processes creating and maintaining fluvial landforms and the ecological community is supported by ecological theory (Gregory et al. 1991, Montgomery 1999, Goebel et al. 2006).

The conceptual framework developed for the GVC was developed by integrating several elements used in the large body of previous work on fluvial classifications of valleys, floodplains, channels, and hillslopes. A suite of fluvial processes including erosion, sedimentation, lateral migration, incision, and transport behave similarly across regions. This consistency provides a physical basis for developing a classification that can span geomorphically-distinct regions (Montgomery and Buffington, 1997; Brardioni and Hassan, 2006). A partial list of regionalized schemes, landform-specific classifications, and scale-specific studies that address the classification of valleys, streams, and riparian areas that were considered in developing the GVC are listed in Table 1.

For this investigation fluvial processes are grouped into three main spheres, which collectively describe the geomorphic setting of the valley: 1) system energy, 2) hillslope coupling, and 3) lateral confinement. Specific processes such as erosion, sedimentation, lateral migration, vertical incision, and sediment transport can be placed within one of the three spheres as described below.

Table 1: Listing of previous classification efforts and their approaches, applicability and attributes.

Group	Date	Approach	Hierarchical	Scale	Region	Advantages	Constraints
Kellerhals et al.	1976	Observational	NO	Channel	Western Canada	Extensive definitions, connects channel to valley	Relies on several qualitative variables when discussing valley attributes
Schumm	1977	Sediment, stability	NO	Channel	Great Plains	Relates sediment and power to channel form	Relies on qualitative measures
Collotzi	1976	Observational	YES	Multi	Pacific Northwest (Oregon)	Four-level hierarchy, strong definitions, incorporates channel and valley bottom	Regionally specific, ecologically redundant
U.S Department of Agriculture (USDA)	1977, 1978	Observational	YES	Multi	Pacific Northwest	Directly applied to management objectives	Regional specific, does not describe all valley geometries
Frissel et al.	1986	Process	YES	Multi	Pacific Northwest (Washington)	Links several scales, addresses temporal aspect	Not explicit with valley typology
Cupp	1989	Observational	YES	Valley Segment (300 m)	Pacific Northwest (Oregon)	Uses several variables to relate channel to valley	Regional specific, large number of types (19)
Nanson and Croke	1992	Stream Power, Sediment	YES	Channel-valley	US	Connects channel morphology to sediment and floodplain	Restricted to floodplain morphology
Knighton and Nanson	1993	Stream Power	NO	Channel	Australia	Relates energy to geomorphic conditions	Vacillates on the equilibrium nature of multiple channels
Whiting and Bradley	1993	Process	NO	Channel	Pacific Northwest	Explicitly uses process to predict form	Very limited morphological applicability, only Headwaters
Rosgen	1994	Observational	YES	Multi	US-Wide	Very detailed in definition and description	Gives no basis for thresholds
Montgomery and Buffington	1997	Process	NO	Channel	Pacific Northwest	Detailed channel morphology, forced and intermediate forms	Regional specific, not useful in lower gradient rivers
Montgomery	1999	Process	YES	Multi	Pacific Northwest	Connects process to ecological significance	Conceptual, limited quantitative measures
Brierley and Fryirs	2005	Process	YES	Multi	Australia	Addresses several major geomorphic processes	Subjective in its variables
Snelder and Biggs, Snelder et al.	2002, 2005	Process	YES	Multi	New Zealand	Uses a hierarchy of controlling factors	Relies on factors that may not be important in all areas
Church	2002, 2005	Process	NO	Channel	US	Quantifies sediment and channel morphology	Requires fine-scale data to be applied
Flores et al.	2006	Stream Power, GIS	NO	Channel	Western US	Introduced and tested drainage area as means of applying scale	Error introduced when using estimated stream power from drainage area
Jain et al.	2008	Process, Stream Power	NO	Catchment	Australia	Strong connections to sediment and stream power	Regionally-specific results, identifies single threshold

System Energy: Large-scale processes of channel pattern, landform development and orientation, and selection of resilient riparian species, have also been noted in connection with system energy (Hupp and Osterkamp, 1985, 1996; Bendix and Hupp, 2000; Twidale, 2004; Naiman *et al*., 2005; Parsons and Thomas, 2007). In the GVC, energy refers to the hydraulic power available to scour and shape valley bottoms and the channels they contain. Energy is characterized using unit stream power or valley slope as its surrogate. The equation for unit stream power (Bagnold, 1966) is

$$\boldsymbol{\omega = \frac{\gamma Q S}{w}}$$

where $\boldsymbol{\gamma}$ is the specific weight of water (9,810 N/m^3), Q is discharge (m^3/s), and S is the energy slope (m/m) approximated by valley gradient, and w is channel width (m) estimated from appropriate regional hydraulic geometry relationships. The divisions between the energy classes used in the GVC are directly related to the three dominant sediment domains of the fluvial system: 1) source areas of erosion and entrainment, 2) transport reaches, and 3) extensive depositional floodplains (Nanson and Croke, 1992; Montgomery, 1999; Brierley and Fryirs, 2005). The type of material and the method of transport are related to system energy (NRC, 2002).

Slope is often used as an effective surrogate for unit stream power in predicting stream types in mountain drainage basins (Grant et al., 1990; Montgomery and Buffington, 1998; Flores et al., 2006). For this study, valley slopes generated from DEMs were used for distinguishing between valley energy types correspond to widely recognized shifts in hydro-geomorphic processes. For example, valleys steeper than 3-4% slope tend to contain confined step-pool and cascade channels with varying degrees of hillslope coupling. As valley slopes become less than

3-4%, the channel types gradually shift to broader floodplains containing plane bed, pool-riffle, and sandy streams.

In low energy systems with unit stream power values less than approximately 10 W/ m^2, channels does not possess sufficient energy to effectively erode the channel banks or floodplain deposits (Nanson and Croke, 1992; Knighton, 1999), and lateral migration is minimal or very slow. Based on the definition of unit stream power for a channel with a typical discharge per unit width on the order of $1m^2/s$, it follows that a slope on the order of 0.001 m/m (0.1%) corresponds to a specific stream power of ~10 W/ m^2 which has been previously identified as a threshold of stream power that separates floodplain types and behavior (Nanson and Croke 1992).

For high-energy fluvial systems, there is little consensus regarding a significant stream power value but general agreement regarding a channel gradient threshold. Many researchers agree that the shift from plane-bed / pool-riffle type channel morphologies to step-pool / cascades suggests a major shift in fluvial dynamics. This shift often occurs near 300 to 400 W/ m^2 or 3 to 4% channel gradient (Collotzi, 1976; USDA, 1992; Rosgen, 1994; Van den Berg, 1995; Montgomery and Buffington, 1997; Church, 2002; Flores *et al*., 2006; Brardioni and Hassan, 2006; Pyne *et al*., 2007; Wohl *et al*., 2007; Thompson *et al*., 2008). If a unit discharge on the order of 1 m^2/s is assumed as above, a slope of 0.03 to 0.04 m/m would yield a specific stream power of 300 to 400 W/ m^2, which corresponds to the Nanson and Croke (1992) threshold between high and moderate energy floodplains.

Hillslope Coupling: Sediment that is eroded, transported, and deposited as a result of fluvial processes is ultimately derived from the adjacent uplands, albeit potentially from distant upstream areas (Sear *et al*., 2003). Shallow landslides and debris flows are often smaller, occur more frequently, and occur on more varied terrain than deep-seated or fault-driven landslides. The contribution of material to the valley bottom or channel can have significant affects on local erosion and deposition, mechanically damage vegetation and adding to the heterogeneity of surfaces in the immediate area and considerable lengths downstream. For this reason, the GVC addresses the potential for colluvial debris to deposit on the fluvial valley bottom.

Coupling refers to the proximity of the hillslopes to the channel and the likelihood that landslides and debris flows on those slopes may move directly across the valley bottom into the stream channel at the slope base. In coupled settings, the channels and the riparian communities occurring along them may be more influenced by materials transported directly from hillslopes (colluvium) than by materials transported from upstream by water (alluvium). In uncoupled settings, sediment recruitment and transport largely become consequences of erosion of the streambed and banks.

The capacity for a hillslope to exert influence on the riparian system is correlated with both gradient and proximity to the fluvial valley bottom. The stability of a slope helps characterize its capacity to transfer material to the valley bottom and/or channel by methods other than simple surface erosion (USDA, 1992; Williams *et al*., 2000; Benda *et al*., 2007). The valley width is a key control on the probability that the colluvial material will encounter the channel and affect flow. Together these two geomorphic variables can be used to approximate the likelihood of colluvial debris being generated on the hillslope and depositing on the fluvial valley bottom (Whiting and Bradley, 1993).

A simple measure of the capability for adjacent hillslopes to generate colluvial material is the composition of slope gradients. Three hillslope gradient classes are designated in the GVC by two user defined thresholds: one describing the lower limit of unstable slopes and the other gradient above which shallow landslides and debris flows are common (Whiting and Bradley, 1993; Montgomery, 1999; Clarke and Burnett, 2003). Default values of hillslope angles used in the GVC were selected based on a review of several previous studies and recognized classifications. Previous work supports a lower threshold of 30% (Collotzi, 1976; Cupp, 1989; USDA, 1992). An upper threshold of ~70% is more uncertain, in part, because regional values of hillslope stability are so widely varied.

The colluvial potential of a valley is simplified into a single value termed the “coupling statistic” (Coup_stat), following Whiting and Bradley (1993). The equation below shows how the length of potential colluvial input deposits (Dr) is related to the “un-channelized valley width” (width of the *total* valley bottom – width of the bankfull channel). The “*# of ‘steep’ sides*” is used to treat the valley as having two, independent hillslopes, each with influence.

$$Coup\_stat = \frac{(\#\ \mathrm{of}\ '\mathrm{steep}'\ \mathrm{sides}) * (\mathrm{Dr})}{\mathrm{Wv} - \mathrm{Wbf}}$$

Lateral Confinement: In most situations, the channel occupies only a portion of the valley bottom at bankfull stage. The un-channelized valley bottom is subject to becoming incorporated as active channel if lateral migration occurs. Lateral confinement can affect the development of extensive floodplains (Dodov and Foufoula-Georgiou, 2005) and the dynamics of floods, which are important considerations for riparian extent, biological composition, and ecological function (Hupp and Osterkamp, 1985, 1996; Bendix and Hupp, 2000; Quinn *et al*., 2000). The concept of lateral confinement has the advantage of scaling to the size of the system.

A common approach to examining confinement is to apply thresholds related to meander geometry advanced by Leopold and Wolman (1960), Leopold and Langbein (1966), Ferguson (1973, 1979), and others. Much of the work on meander geometry originates with the sine-generated wave introduced by Langbein (1966). This graphical approach mirrors concepts behind the adjustments in channel form as meanders and the sine-generated wave minimize the changes in direction and work performed by the system. Several mathematical relationships have been identified that describe the geometric shape of meandering channels and their cross sectional profiles (see Leopold and Wolman, 1957, 1960; Langbein, 1966; Leopold and Langbein, 1966; Williams, 1986; Hagerman and Williams, 2000; Soar and Thorne, 2001).

Two distinctive geometric characteristics of meanders of any origin are: 1) amplitude and 2) wavelength. Williams (1986) examined dozens of ways these and other attributes of channel meanders can be related to each other. Here we use the relationship between meander amplitude and wavelength, herein referenced as *A* and λ, respectively, to identify the threshold at which free-lateral adjustment becomes impeded. This relationship assumes a sine-generated curve (Leopold and Langbein, 1966). Hagerman and Williams (2000) developed a third-order polynomial to calculate the meander amplitude using λ as the independent variable:

$$A = \lambda(6.0625\varphi^3 - 5.1279\varphi^2 + 2.509\varphi + 0.0005)\,.$$

The term φ is (*P - 1*)/*P*, where *P* is the sinuosity. Wavelength cannot always be directly measured; however, but Soar and Thorne (2001) proposed a relationship between bankfull channel width *Wc* and λ based on a large meta-analysis of meandering rivers around the world: λ = 12*Wc*. The polynomial relationship for *A* above can be rearranged to calculate meander belt width (*B*) by substituting 12*Wc* for λ and adding a channel width:

$$B = (12(6.0625\varphi^3 - 5.1279\varphi^2 + 2.509\varphi + 0.0005) + 1)W_c$$

The additional channel width accounts for meander amplitude being measured between two points located in the center of the channel. For confinement, interest is in the outer edges of the channel.

To arrive at a threshold for the minimum unconfined valley width, the conventional definition of a meandering channel as maintaining a minimum P of 1.5 is used (Leopold and Wolman, 1957, 1960; Van den Berg, 1995). It follows from the relationship for B that the threshold for the minimum valley bottom width that can contain the belt width of a meandering channel with a P of 1.5 is approximately $7Wc$. To the extent that sine generated wave meander geometry is a reasonable approximation, this value is scale independent and transfers between regional physiographic boundaries.

A second threshold is identified for riparian settings in highly-confined situations. Lateral migration of the channel is not an option for these narrow valleys, but confinement is nonetheless still important. A shift in channel morphology from single thread to braided has been shown to be related to channel slope, sediment load, and variable discharge (Leopold and Wolman, 1957; Schumm and Khan, 1972; Fredsoe, 1978). This results in a dramatic change in the width- to-depth (W/D) ratio from ~25:1 for single-thread channel while braided channels are often twice that, at 50:1 (Fredsoe, 1978). A valley width index (VWI) (the ratio of valley width to bankfull channel width) of 2 has been highlighted as a threshold separating systems where there is no ability to maintain long-term depositional features. Transient floodplain features are critical in these habitats, but a characteristic alluvial floodplain does not develop. This value is supported by several established regional valley classifications (Collotzi, 1976; Cupp, 1989; Rosgen, 1996; O’Connor and Watson, 1998).

Hierarchical Organization

The three spheres of processes create a strong foundation for the development of a fluvial classification as they describe the significant forces which shape the valley, floodplain, and channel. The balance between explanatory power and complexity is a common struggle in scientific investigations. Each process can be quantified with a simple surrogate from readily available GIS-data layers with relatively simple, well-known procedures. Even within the GVC, a hierarchy exists for the geomorphic variables (see Figure 1) and is critical in the determination of the final valley classes (Table 2). The interplay between the three processes highlights the interconnectedness of the fluvial and hillslope components (Brierley and Fryirs, 2005; Wohl *et al.*, 2007).

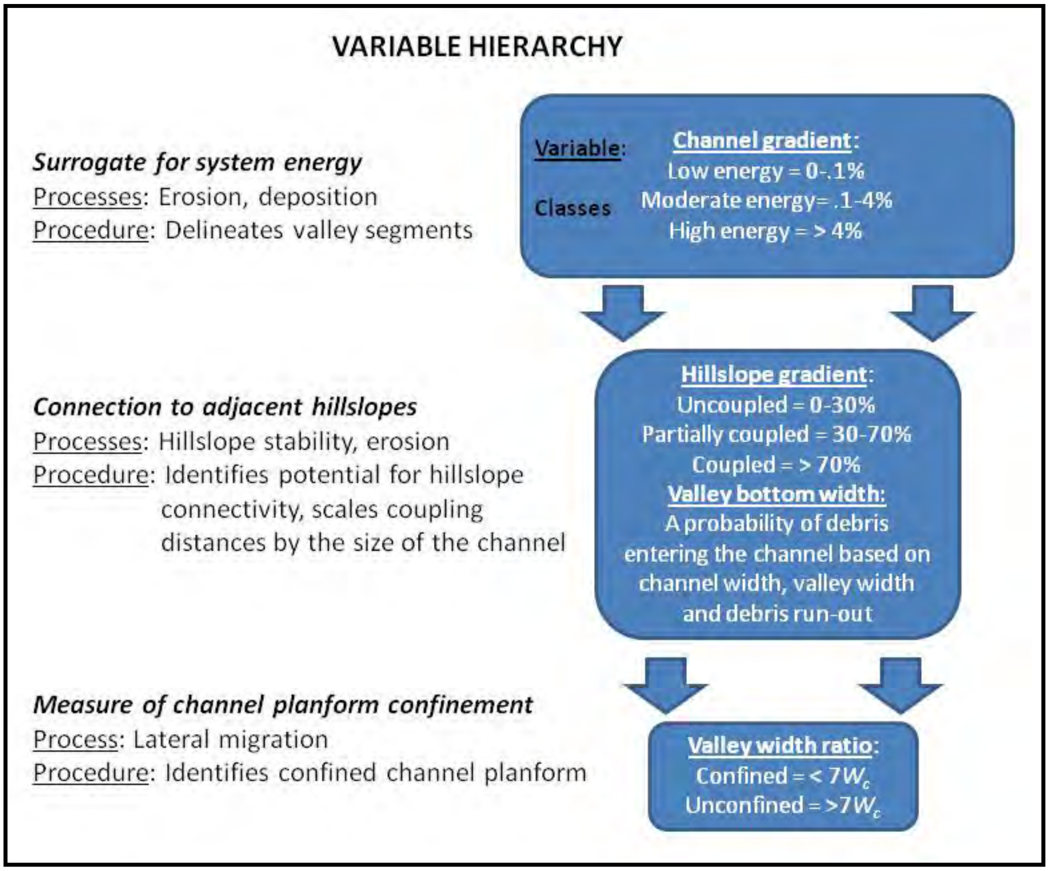

Figure 1: The variable hierarchy used to constrain processes and landforms at successively finer scales.

A direct comparison between the approach to valley classification presented by Rosgen (1994, 1996) and classes of the GVC defined in Tables 2 is warranted as Rosgen's approach has been widely used for the last fifteen years in public and private land management (Table 3).

Table 2: Valley classification names and attributes of the GVC.

Valley Class Name	Energy / Valley Gradient	Valley Bottom Width / Coupling / Confinement	Hillslope Gradient	Energy Potential
Headwaters	> 4%	$< (2 L_D + W_{BF})$	Both > 30%	High
High-energy Coupled	> 4%	$< (2 L_D + W_{BF})$ or $< (L_D + W_{BF})$	Both or at least one > 30%	High
High-energy Open	> 4%	$> (2 L_D + W_{BF})$	Both or at least one > 30%	High
Moderate-energy Confined	0.1-4%	$< 7 W_{BF}$	Variable	Moderate
Moderate-energy Unconfined	0.1-4%	$> 7 W_{BF}$	Variable	Moderate
Canyon	Variable	$> 3 W_{BF}$	> 70%	Moderate to High
Gorge	Variable	$< 3 W_{BF}$	> 70%	Moderate to High
Glacial Trough**	< 4%	$> (2 L_D + W_{BF})$	~ 10-% initially steepening to > 30%	Moderate to Low
Low-energy Floodplain	< 0.1%	$> 7 W_{BF}$	Generally < 30%	Low

L_D – length of debris runout W_{BF} - width of channel at bankfull stage

** Defined as valleys with the given characteristics, lying above the elevation of the most recent glacial activity

Table 3: A comparison between GVC valley classes and Rosgen (1994) valley types.

GVC				Rosgen (1994)			
Valley Class	Channel Gradient	Confinement	Hillslope Gradient	Valley Type	Channel Gradient	Confinement	Hillslope Gradient
Headwater	> 4%	Confined	Steep	1	> 2%	Confined	Steep
High-energy Coupled	> 4%	Confined	Steep	1	> 2%	Confined	Steep
High-energy Open	> 4%	Unconfined	Steep	2, 3, 6, 7	> 2%	Moderately confined	Moderate-Steep
Moderate-energy Confined	0.1 - 4%	Confined	Low-Steep	2, 3, 6, 7	< 4%	Confined-Moderately confined	Moderate-Steep
Moderate-energy Unconfined	0.1 - 4%	Unconfined	Low-Steep	6, 9	< 2%	Moderately confined	Moderate
Canyon	Variable	Confined	Steep	4	< 2%	Confined	Steep
Gorge	Variable	Confined	Steep	4	< 2%	Confined	Steep
Glacial Trough	< 4%	Unconfined	Moderate-Steep	5	< 4%	Unconfined	Moderate

Low-energy Floodplain	< 0.1%	Unconfined	Low-Moderate	8, 9, 10, 11	Low	Unconfined	Low

GIS Application of the Geomorphic Valley Classification

A suite of spatial techniques for analyzing river networks was developed parallel to the GVC in a robust computerized mapping environment. The code, scripts and spatial tools from ESRI ArcGIS v. 9.3 were manipulated in a protocol that yields the information about a given valley necessary to classify it using the GVC approach. Unique and novel portions of the coding are described below.

Data inputs: The single data input into the GVC Mapping Protocol is a 10-m resolution digital elevation model (DEM). Regional parameters are defined by the user as prompted, including hillslope stability analysis, elevation of glacial influence, and regression equation coefficients for estimating bankfull channel width (see Faustini et al. 2009). Defaults are provided within the Protocol, but are based on regional mean values.

Three spatial analysis modules: The spatial analysis of a given river network in the GVC can be broken down into three fundamental spatial analysis modules. The first module identifies the initial valley segments using a series of steps to create and smooth channel gradient into a measure of valley gradient. The raster cells are then reclassified into one of three gradient classes (High, Moderate and Low-as defined in Table 2) and grouped together. Groups of cells less than a user defined minimum are eliminated and the adjacent groups “extend” or “grow” into the empty network spaces. This gives a minimum length for valley segments. We recommend nothing smaller than 200m, as channel scale processes become more dominant below that distance and are more difficult to predict in GIS with current data limitations. The resulting segments are used throughout the remaining analysis as the defined valley segment network.

The second module examines the topographic and hydrologic attributes of each valley segment individually. A Q100 discharge (100-yr peak flow) is estimated for every valley segment using the equations from the USGS National Stream flow Statistics Program (http://water.usgs.gov/osw/programs/ nss/summary.html). This module employs Manning's equation to the DEM in order to extract the flood elevation that corresponds to the volume of water predicted by the Q100 estimation. The flood elevation is used to create the 'hydrologic' valley bottom. It is also used to limit the extent to which the landscape is analyzed to identify breaks-in-slope.

Following the NRP riparian definition, the Q100 is the default riparian boundary, but we wanted to incorporate the unique topography into a more descriptive valley extent. The horizontal area within a vertical distance above the channel (related to Q100) is used to define the area over which to examine breaks-in-slope. This extent is mirrored by a lower extent, defined as a factor of bankfull channel width, essentially creating an annulus (Figure 2) where the analysis of the break-in-slope occurs.

The *lowest* values of the curvature are the *most* concave cells (darker red). The 20% most concave cells (lowest values) within the area of analysis are extracted and their elevation above the channel is averaged and used to create a second valley bottom layer is created. These two layers are combined by reporting only the areas where they overlap, the *hydro-geomorphic* valley bottom (Figure 3). This is the ultimate representation of the valley extent used in the final classification.

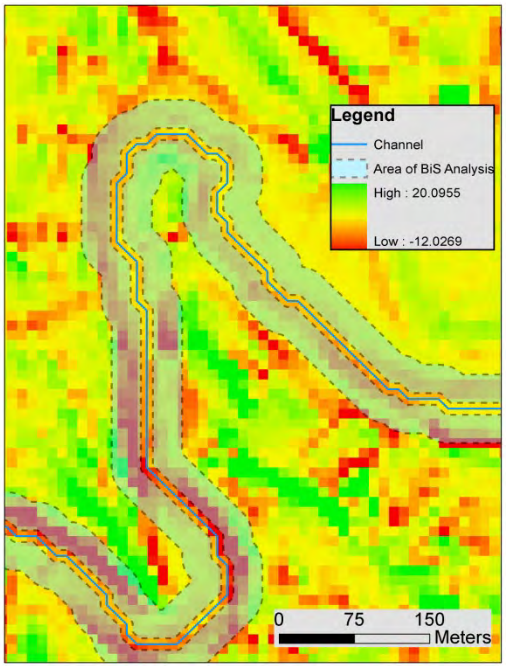

Figure 2-The area used to analyze the topography for breaks in slope

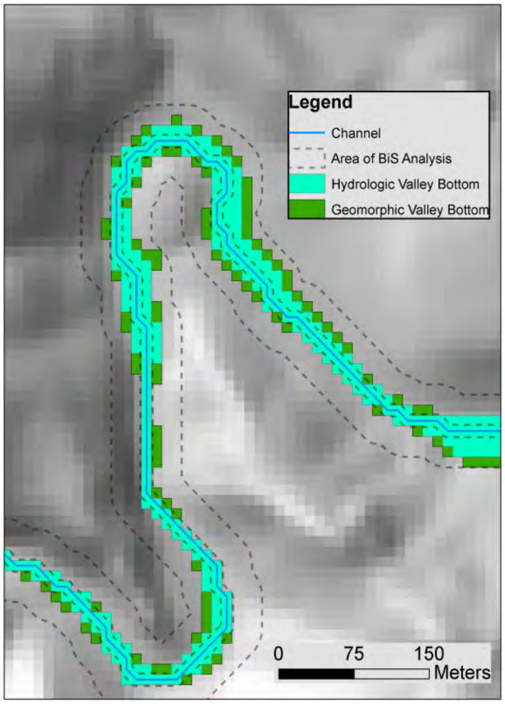

Figure 3-Overlay of hydrologic and geomorphic valley bottoms

The final module examines the adjacent landscape for its potential influence on the valley bottom through confinement and colluvial inputs. The adjacent slopes are categorized by gradient into three classes (Steep, Moderate, Low). All three affect confinement of the channel, while only Steep and Moderate have the potential to contribute colluvial material and are employed as the "'steep'sides" variable from Equation 4. It would be very difficult to examine hillslopes using cross-sections oriented transverse to the channel to obtain the slopes necessary for categorizing gradient. Instead we use a composition of the slopes, as a percentage of the slope in each of the three gradient classes. The proportions then give a slope its final category.

The second and third modules described above are completed automatically following the proper designation of files and parameter inputs. It is important to note that the valley bottoms are created and hillslopes analyzed on *a segment by segment* basis which makes this a more robust methodology.

Final Classification

The final classification is performed when all spatial layers have been created. Two additional calculations are completed to give values such as the ratio of valley width the channel width (used to determine confinement) and the coupling statistic. The entire protocol is set up in a series of three main steps: 1) create the initial valley segments, 2) manually edit inconsistencies in the valley segments and 3) create the valley bottom, analyze the hillslopes and classify the valley. Step 1 is a tool built in ArcGIS 9.3 and Step 3 is a Python script run within ArcMap as tool. Step 2 is a necessary manual step that allows the user to become familiar with the outputs to identify and address any errors.

Field Testing

Three study regions in three western states were chosen to test the GVC; two were located on USFS land and one was located on a combination of Bureau of Land Management (BLM), State of Wyoming, and private land. Forty-two field sites were visited: 17 in Arizona, 14 in Colorado, and 11 in Wyoming. Five ecoregions were encountered, each characterized by different climates, geology, and supporting unique species pools. Sizes of the watersheds examined were varied to capture differences inherent with gross size of the system (Table 7.2). Vegetation was dramatically different between the three study regions, ranging from desert to deep coniferous forests. Flow regimes also varied dramatically; for example, Cherry Creek in Tonto NF records its highest average monthly discharge in February and lowest in June, while the Fraser River in the Arapahoe NF records the exact opposite trend (USGS, 2009).

Table 4: Study area basin parameters showing the major stream name, USFS unit, drainage area, eco-region, vegetation, and landscape character.

Stream Name	Forest	Drainage Area	Eco-region	Upland Vegetation	Landscape
Green River	N/A	~19400 km^2	Temperate Desert	Sagebrush	Rolling
Black's Fork	N/A	~9500 km^2	Temperate Desert	Sagebrush	Rolling
Pinal Creek	Tonto	515 km^2	Tropical / Subtropical Steppe	Spruce/Pine forest, Semi-arid shrubs	Mountain headwaters, rolling
Pinto Creek	Tonto	482 km^2	Tropical / Subtropical Steppe	Arid - Semi-arid shrubs	Rolling
Cherry Creek	Tonto	720 km^2	Tropical / Subtropical Steppe; Tropical / Subtropical Regime Mountains	Spruce/Pine Forest, Semi-arid shrubs	Mountain headwaters, rolling
Williams Fork	Arapaho	370 km^2	Temperate Steppe Regime Mountains	Spruce/Pine/Fir forest	Mountain
St. Louis Creek	Arapaho (Fraser Experimental Forest)	96 km^2	Temperate Steppe Regime Mountains	Spruce/Pine/Fir forest	Mountain
Fraser River	Arapaho	78 km^2	Temperate Steppe Regime Mountains	Spruce/Pine/Fir forest	Mountain

N/A – not applicable

The under-estimation of slope values in GIS has been well-documented (Brardioni and Hassan, 2006). This investigation proved no different, as field values were higher than GIS values by 160, 83, and 48% for the Wyoming, Arizona, and Colorado study regions, respectively. Of the 42 field sites, 17 had channel gradients misclassified in GIS that led to the designation of a different energy regime. The under-estimation of gradient in GIS was common throughout all study regions and prompted the lowering of the GIS threshold between High- and Moderate-energy from 4% to 3%.

For the valley bottom width variable, the discrepancies between field and GIS values were the lowest of any variable. Differences were 27, 58, and only 8% for Wyoming, Colorado, and Arizona study regions, respectively. This was not expected and suggests that the measure of confinement is the most robust geomorphic variable for this data set. HEC-GeoRAS comparison to GIS data showed vast differences in the fluvial valley bottom extent identified by the modeled 100-yr flood extent and the GIS method used in the GVC. As predicted, the GVC method for identifying valley extent preformed better at the 1-m scale compared to the 10-m scale when using the *Q*100 extent from HEC-GeoRAS as the “true” extent.

Brardioni and Hassan (2006) also found that hillslope angles measured in GIS were routinely under-estimated in steep areas, and over-estimated in low gradient, depositional environments. As expected, the least amount of difference occurred in the Colorado study region. Here slopes were generally steep, which means that the contour lines from which the DEM was digitized and subsequently interpolated between are closer together. This equates to more data points within a given area and thus a more accurate value. The values in the Arapaho NF in Colorado had the differences of 20 to 50% between GIS and field-observed values. Hillslopes in Wyoming were the most poorly represented with disparities between field-observed values and GIS of values of 150 to 250%.

The poor identification of vertical hillslopes was most apparent in Arizona, where several Canyons were misidentified because they lacked the excessively steep hillslopes characteristic of

Canyons. Vertical and nearly vertical hillslopes are the key delineative criteria for Canyons and Gorges, along with a narrow valley bottom width. Lacking accurate hillslope classification likely led to the misclassification of Canyons as Moderate-energy Confined and High-energy Coupled valley classes.

In general, the testing, results, and subsequent analysis of the GVC classification and the GIS procedures illuminated some interesting successes and shortcomings. Qualitative field observations of the GVC generally succeed in identify logical breaks in geomorphic and ecological function at the valley scale; however, the quantification for the classification and the GIS procedures was not statistically robust, owing to the limited number of “sites” (42) in the field spread between 3 study regions, and 22 cross sections in HEC-GeoRAS. Despite these limitations, general patterns and shifts in valley setting could still be detected using the GIS-based approach.

Summary

The Geomorphic Valley Classification is a framework for stratifying fluvial riparian systems that aids in regional and landscape scale management decisions. The process-informed classification is simple, requires minimal data inputs, and is executed using a common GIS platform. The thresholds separating process groups are supported by detailed hydrologic, geomorphic and hydraulic research. In particular, the GVC builds upon work by Montgomery and Buffington (1993), Nanson and Croke (1992), Whiting and Bradley (1993) to create a widely applicable, process-based hierarchy to classify river networks. The GIS procedure has been developed as a semi-automated protocol for creating the necessary data layers to accurately assess the geomorphic structure of river networks. The open framework prompts users to specify local or regionally-calibrated values for thresholds such as the lower limit for unstable hillslope angle, debris run-out length, and a contributing area for valley initiation. Additional details on the GVC methodology are presented in Carlson (2009).

The GIS procedure was developed as a mix of accepted approaches, modified existing methods, and novel means of extracting geomorphic information from readily available DEM data. A novel approach to delineating fluvial valley bottoms was introduced at the center of the GIS procedures. The method for quantifying valley bottom width identifies changes in gradient of the land adjacent to the channel. This method extends a similar approach to evaluating relative surface slope and elevation as a method to characterize landforms (the TPI from Jenness (2006)) by adding components of hydrology to constrain the measurements around the channel.

Errors begin to accumulate immediately in GIS when using remotely-sensed data because the input layers are approximations of true values. Errors are further introduced by GIS procedures and smoothing algorithms. An analysis of error propagation was not performed in the study but it is understood that significant differences between remotely-sensed values and field values exist in some instances. For example, results indicate that the present GIS procedures do not perform satisfactorily when identifying channel gradients less than 0.1%.

With a limited number of test sites (42 field and 22 HEC-GeoRAS), the relatively large number of valley classes (9), and the amount of climate, geologic, and vegetation variability among the six ecoregions does not allow accurate estimates of misclassification rates. A larger data set of verified valleys would permit the examination classification strengths and perhaps illuminate problems with threshold values or measurement techniques.

The regions of the western US where the GVC is hypothesized to be the most accurate are in areas with high relief and wider valley bottoms. High-relief areas will provide a more accurate measure of channel and hillslope gradients, and wider valley bottoms limit the effect of horizontal data resolution on smaller valleys. In general, the technique performed relatively well in Colorado in the estimation of slope and hillslope angles as compared to other regions.

Regions of expected poor performance include extremely flat systems, with gradients less than 0.1%. The current measurement techniques cannot examine slopes below this value. Land highly dissected with canyons or arroyos are also expected to have higher misclassification rates because the valley morphology is often too narrow for the horizontal resolution of the 10-m DEMs. Highly-dissected regions also have the issue that the hillslopes may not be 150-m long, and land that drains to a different channel or the inclusion of a nearby valley bottom could skew the hillslope categorization. LiDAR or field reconnaissance would be necessary to identify canyons or arroyos with widths less than 10 m.

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Appendix D

Validation and Application of Flow-Ecology Methods for Trout

VALIDATION AND APPLICATION OF FLOW-ECOLOGY METHODS FOR TROUT

CONTENTS

INTRODUCTION

The purpose of this section is to review application of flow-ecology relationships for trout from (Wilding and Poff 2008) for the upper-Colorado basin. Specifically, how valid are the relationships when evaluated against independent datasets? This report focuses on Method 3 from (Wilding and Poff 2008), which provides a categorical rating of low flow suitability (Table 1).

Table 1 Method 3 reproduced from (Wilding and Poff 2008) for trout in Rocky Mountain streams. 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 a percent of mean annual flow.

Rating	Summer low flow (% of mean ann. 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.

VALIDATION OF FLOW-ECOLOGY RELATIONSHIPS

Validation of the Method 3 flow-ecology relationship was investigated using data from (Nehring 1979). In that report (Nehring 1979) compared the Montana method (Tennant 1976) to minimum flow recommendations based on R2-Cross and a pre-cursor method to PHABSIM (IFG4 is the hydraulic model implemented without a biological model). Sites included a wide range of Rocky mountain streams, with sixteen sites surveyed using R2-Cross and 14 using IFG4. Colorado basin sites included several on the Frying Pan, Williams Fork and Saint Louis. The minimum flows were based on similar objectives across methods (minimum water depths and/or velocities), with IFG4 producing minimum flow estimates based on more intensive sampling over the stream length and over time.

The R2-Cross method (single cross-section) estimates flows required to maintain depths sufficient to keep the largest fish wet as it swims through a riffle. Minimum flows produced using R2-Cross assessments averaged 28.4% of MAF, and IFG4 assessments averaged 27.9% of MAF (standard deviation 10.5% and 10.8% respectively). The surveyed minimum flows (R2-Cross and IFG4) were compared to the flow calculated as 25% of MAF for each site that was assessed by (Nehring 1979), and this relationship is presented in Figure 1. There is variability in the relationship with survey estimates, but the Method 3 approach provides a good approximation. The 25% of MAF distinguishes flows that “may severely limit trout” from flows that “may occasionally limit trout” (Table 1), so it seems appropriate for minimum flows based on the water depth required to keep a fishes back wet in riffles. The 25% threshold has appeared in many publications, which may have a common origin (earlier publications by Thomas Wesche).

The depth criteria used for R2-Cross, which dictated the recommended flow in most cases, was based on a sliding scale[1]. Therefore the indifference shown by the Method 3 to absolute stream size was effectively built into the R2-Cross assessments as well. (Nehring 1979) anticipated that R2-Cross would not be successful in predicting trout performance and recommended it for flow prescriptions on lower-value streams. The same study achieved good correlations with trout biomass using habitat suitability at MAF, using absolute depth-velocity criteria (i.e. not sliding scale) modeled in PHABSIM (termed IFG3 at the time), and he recommended this method for flow prescriptions on high-value fisheries.

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[1] Average riffle depth 1% of stream width for streams 20-100 feet wide, and 0.2 feet for smaller streams. Based on height of the largest fish present from tip of dorsal to lowest portion of body cavity.

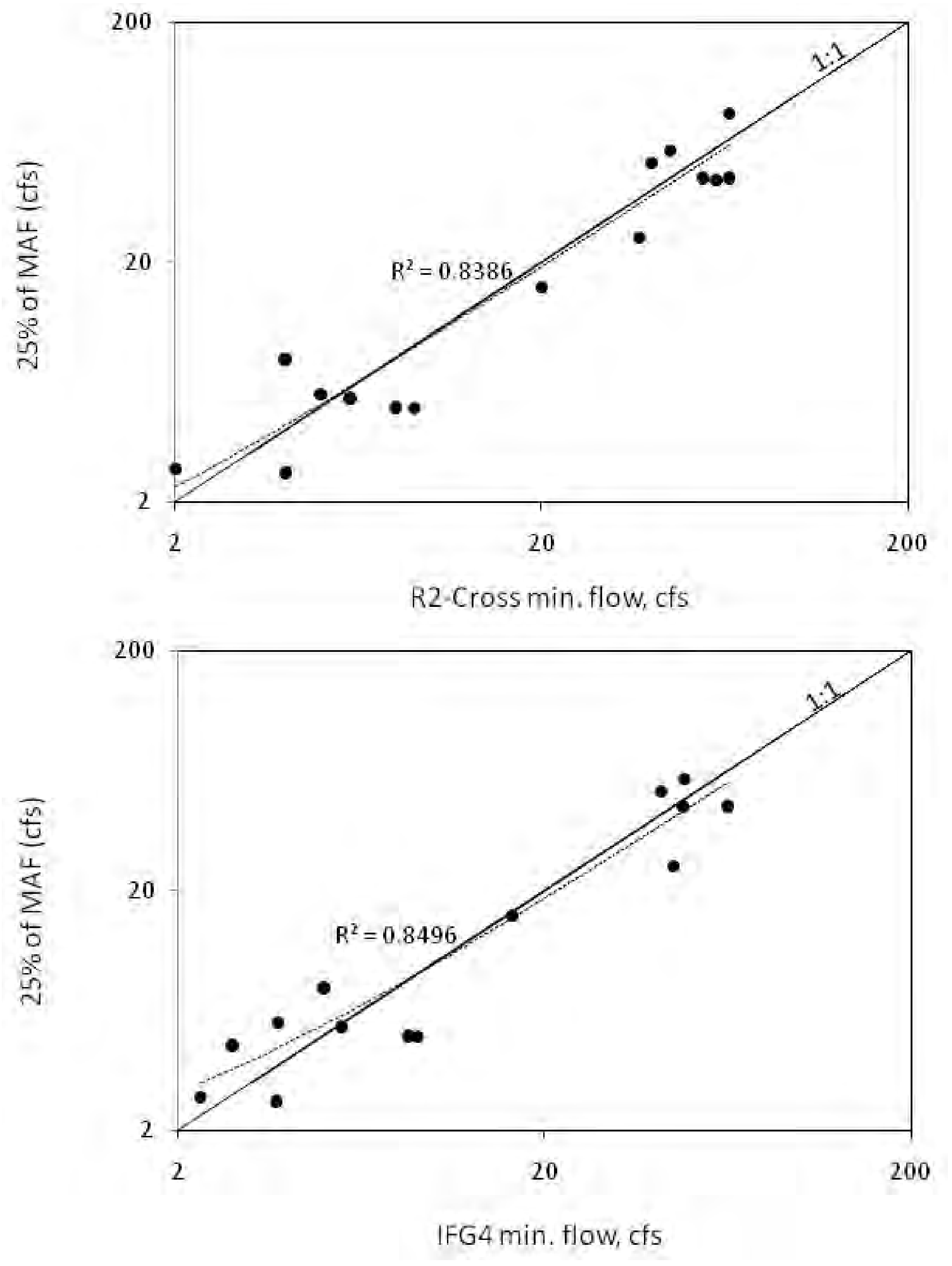

Figure 1 The relationship between 25% of MAF (threshold from Table 1) and minimum flows estimated using R2-Cross (upper plot) and a more intensive equivalent (IFG4 using similar depth targets – lower plot). The relationship (dashed line labeled with R^2 value for linear least squares) approaches the 1:1 relationship (solid line) expected if values were a match.

Additional validation of Method 3 was achieved by comparing predictions to Instream Flows from the Colorado Water Conservation Board (Water Division 5 – Upper Colorado River Basin). These allocations are typically based on R2-Cross assessments, but the final allocation likely incorporates other considerations, including water availability. There are additional complications in this comparison. To start with, the instream flows apply to long sections of stream, so it is not apparent whether any specific StateMod node corresponds to the survey point or points. There may also be several separate allocations that are not summed in this comparison. Automated GIS methods were not successful in matching Instream Flows to StateMod nodes for various reasons, so nodes were manually matched to instream flows. A subset of sites were randomly selected from the Division 5 summary (random selection stratified by 4 flow groups), producing 12 successful comparisons from 33 attempts (comparison not possible where nodes do not correspond or location descriptions were insufficient). The Method 3 prediction was again represented by the Category 2/Category 3 demarcation of 25% of natural MAF. Where instream flows varied seasonally, only the Instream Flow value that overlapped with the August-September period was used, as this is the period assessed using Method 3.

There is more variability in the relationship between the Instream Flows and Method 3 (Figure 2), compared to the Nehring results (Figure 1). But the Instream Flows approach 25% of MAF on average, as indicated by the regression line that lies close to the 1:1 line, and therefore provides further support for Method 3.

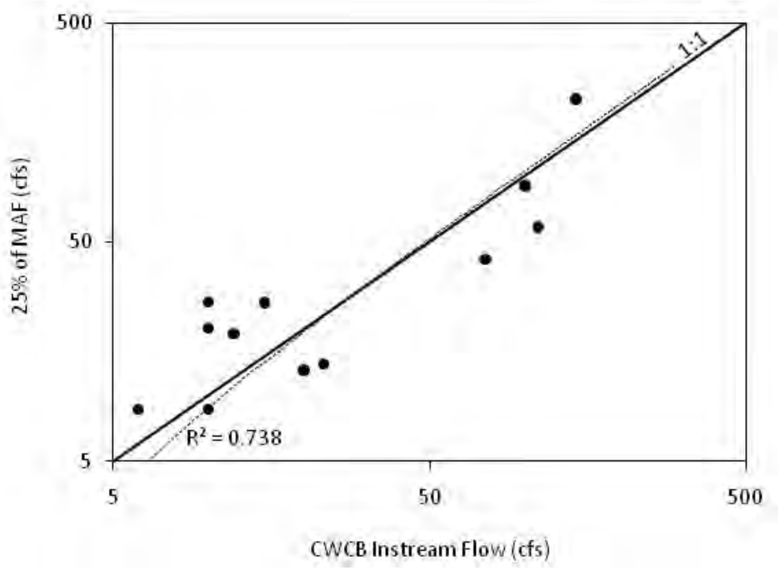

Figure 2 The relationship between 25% of MAF (threshold from Table 1) and Instream Flows allocated by the Colorado Water Conservation Board. The relationship (dashed line labeled with R^2 value for linear least squares) approaches the 1:1 relationship (solid line) expected if values were a match.

CUTTHROAT TROUT

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). Self-sustaining populations of indigenous cutthroat trout (lacking hybrids) are typically isolated from introduced trout by instream barriers such as diversion structures (Young 1995, Young et al. 1996). The distribution of cutthroat trout is no longer the realization of suitable temperature and flow regimes, but instead tolerable refuges from introduced trout. This poses a more complex problem for predicting distribution. Direct mapping of conservation populations offers a better alternative to statistical modeling, given the small number of remaining populations and their well documented occurrence (Young et al. 1996).

The next question then is what relevance does Method 3 hold for these populations? Cutthroat trout are confined to small, high-elevation streams where they can avoid introduced trout, and these are places where both flow magnitude and stream temperature are expected to be marginal. This was demonstrated by (Harig and Fausch 2002), with a high chance of failure of translocated cutthroat trout in small streams with cold summer water temperatures. Several studies have demonstrated the greater proportion of mean flow required to maintain trout habitat in smaller streams (Hatfield and Bruce 2000, Lamouroux and Jowett 2005). Therefore, it is reasonable to assume that stream flow is a limiting factor for remaining cutthroat trout populations. It is worth clarifying that flow is not the primary limiting factor - clearly barriers to invasion from introduced trout are of primary importance for persistence of conservation populations. Rather it implies that flow acts to constrain populations of cutthroat trout in the small streams that lack introduced trout.

The study by (Binns and Eiserman 1979), upon which Method 3 was based, certainly includes many small streams (MAF not provided for all sites, but 20% of sites were < 3m wide during summer), so is at least relevant. The study by (Nehring 1979) has already provided some validation of the 25% of MAF threshold for smaller streams that support cutthroat trout, based on R2-Cross predictions (see the section Validation of flow-ecology relationships). What's missing is population level validation specifically for cutthroat trout in small streams. But effort may be better invested in site specific investigations, and there are several reasons for this. The values and objectives associated with conservation of cutthroat trout differ to management of recreational fisheries. The flows required to ensure that a population persists are likely less than the flows required to support a gold medal fishery, but flow is more severely limiting in the small streams where cutthroat are now confined. (Young 1995) cited a Wyoming study where most populations of cutthroat persisted in streams less than 0.85 m^3/s (mean annual flow, cf. Figure 2). Additionally, the limited distribution of cutthroat trout reduces the value of regional methods - (Young et al. 1996) reported 9 “conservation populations” and ~70 compromised populations in the upper Colorado basin. Method 3 will at least provide a basin level picture of where reduced flows coincide with cutthroat trout populations and aid in prioritizing such assessments.

RECOMMENDATIONS

The area of application was not evaluated in this report as the Colorado Division of Wildlife has provided advice on this matter. This report instead focused on validation of Method 3 using physical data including R2-Cross surveys, more intensive surveys pursuing similar flow targets and Instream Flows that are in some way based on R2-Cross surveys. These support Method 3 as providing an adequate approximation of R2-Cross predictions at least for the purposes of basin wide planning assessments. The method is not expected to provide an adequate substitute for site-specific flow prescriptions, particularly where cutthroat trout populations of conservation value are concerned, or valued recreational fisheries.

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Appendix E

Warm Water Fish Methods for the Watershed Flow Evaluation Tool

Warmwater Fish Methods for the Watershed Flow Evaluation Tool

A report to the Non-Consumptive Needs Committee of the Colorado Basin Roundtable

February 2011

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Contents

Summary and Recommendations

For the Watershed Flow Evaluation Tool Pilot Study (WFET; CDM et al. 2009), Wilding and Poff (2008) developed methods for evaluating risk to warmwater fish species resulting from water development and management that changes the timing and quantity of river flows. This current report incorporated feedback on the concepts presented in the 2008 report and more analysis was done, leading to modification of WFET methods for warmwater fish (Wilding and Poff 2008). The two most significant modifications are: (i) we are no longer recommending a Method 10 from Wilding and Poff for evaluation of the listed endangered Colorado pikeminnow; rather, we are incorporating U.S. Fish and Wildlife Service flow recommendations into the WFET, and (ii) the method for non-listed, at-risk warmwater fish species (flannelmouth sucker, bluehead sucker, and roundtail chub) was modified to a method based on data for the two suckers but not roundtail chub. Also, this modified sucker method was refined to apply only in specific geomorphic settings, was adjusted to eliminate potential conflating for flow effects and non-native predatory fish effects, and was validated using independent data..

Flow targets for endangered fish are drawn from documents available through the Upper Colorado River Recovery Program. For the Colorado River, the primary reference is the Programmatic Biological Opinion (PBO) for the 15-mile reach between Palisade and Grand Junction as well as supporting documents, especially Osmundson's 2001 report on flow regimes for restoration and maintenance of sufficient habitat to recover endangered Razorback Sucker and Colorado Pikeminnow in the upper Colorado River.

In this report we analyzed gaged flows at one location (Palisade, gage 09106150) compared to a subset of the 15-mile reach flow recommendations. This analysis of gaged flows for the period of record (water years 1991-2010) illustrates that flow recommendations are frequently not attained at this location. We present the results of this analysis as a relatively simple approach to using output from the state's surface water flow model (StateMod) to readily compare how multiple water management scenarios perform with respect to PBO flow recommendations. However, we do not suggest that this approach is a definitive statement as to how water management on the main stem is doing to achieve endangered fish flows in the 15-mile reach.

For bluehead sucker and flannelmouth sucker, a revised function is presented for evaluating the biological effects of long-term changes in flow:

$$\%\ \text{maximum native sucker biomass} = 0.1026 \times \text{30-day min flow}^{0.3021}$$

where '30-day minimum flow' is a running mean) calculated over the summer-autumn flow period (1 July to 30 November) for each year, then averaged over the study period (1975-2005).

In this manner, biomass is estimated for both baseline (natural) conditions and a managed scenario (typically ‘current’ at a minimum). Percent reduction in biomass is then calculated as:

% reduction in biomass = (baseline – current) / baseline)

Risk classes based on expert recommendations are: low risk (0-10% reduction in biomass), minimal risk (10-25% reduction), moderate risk (25-50% reduction), and high risk (50-100% reduction).

Flannelmouth and bluehead sucker are warmwater fish, so it is important that the method is not applied where cool water temperatures may override the flow response. Therefore the sucker method should only be applied at nodes below 7,000 feet elevation. More specific limits can be specified for the mainstem of the Colorado River - Radium at 6,850 ft (downstream of USGS 09058030). Likewise, on the Roaring Fork River a specific upstream limit at the Frying Pan confluence is recommended (6,590 ft). Within this temperature envelope, application of the sucker method should be further constrained to exclude low energy reaches (channel slope <0.1%) to focus on reaches with more suitable habitat (rocky substrate).

The use of low flows to indicate sucker response was supported by spatial validation analysis using independent catch data from the Colorado basin. Sucker monitoring data from the San Juan River demonstrated that sucker populations do not follow inter-annual flow fluctuations. The sucker method therefore describes change in carrying capacity over longer periods, rather than year-to-year variability.

Introduction

The purpose of this section is to review the flow-ecology relationships for warmwater fish that were developed by (Wilding and Poff 2008). Specifically, where in the upper-Colorado basin should the methods be applied and do the relationships hold if validated against independent datasets? This report focuses on Method 9 from (Wilding and Poff 2008), which describes the relationship between low flow and potential biomass of bluehead sucker, flannelmouth sucker and roundtail chub. The flow metric used to describe low flows was revisited for this report using additional flow metrics that were not included in the source document (Anderson and Stewart 2007). The potential to describe the flow-ecology response for all three species using one method was also investigated (compared to 3 equations for Method 9). Alternative methods for Colorado pikeminnow are also reviewed.

Endangered fish of the 15-mile reach

Four species of fish that are listed as Endangered are affected by water management on the mainstem of the Colorado River: Colorado pikeminnow (*Ptychocheilus lucius*), humpback chub (*Gila cypha*), bonytail (*Gila elegans*), and razorback sucker (*Xyrauchen texanus*). However, only the Colorado pikeminnow and razorback sucker are currently known to occur in the 15-Mile Reach. The PBO (US FWS 1999) reviews the critical aspects of the biology and ecology of these species as well as the importance of the 15-mile reach to Colorado pikeminnow and razorback sucker.

Flow-ecology methods for endangered fish in the 15-mile reach

Wilding and Poff (2008) developed a generalized flow-ecology relationship for Colorado pikeminnow that they referred to as Method 10. The U.S. Fish and Wildlife Service (US FWS 1999) presents specific flow recommendations for endangered fish were presented for the 15-mile reach (Figure 1) in the Programmatic Biological Opinion. These specific recommendations

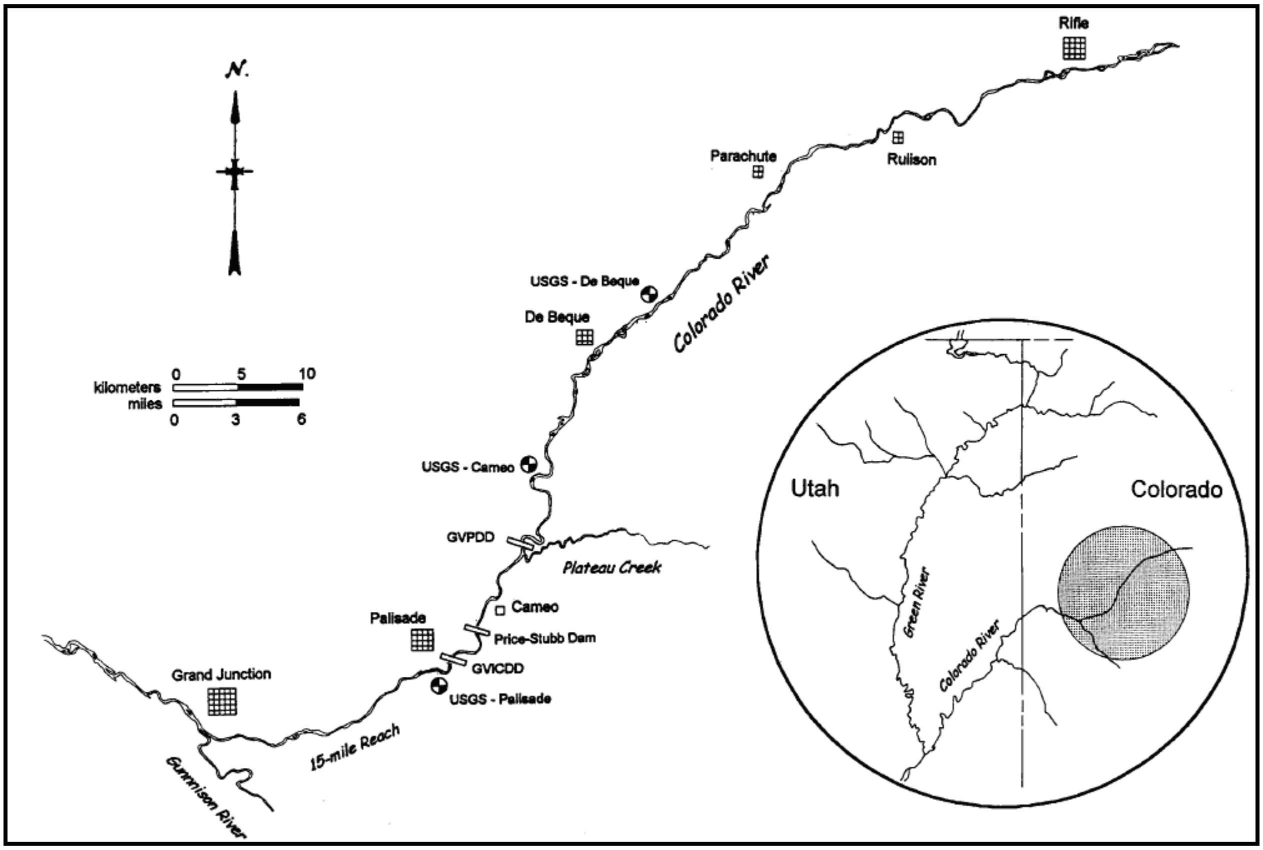

Figure 1 Reproduced from (Osmundson 2001), this map shows the 15-mile reach between Palisade and Grand Junction where endangered fish management is focused, plus the reach extending to Rifle where improved fish passage might restore populations as far upstream as Rifle.

(reproduced in Tables 1 and 2 below) were based on extensive research by scientists with detailed knowledge of these species and their habitats (e.g., Osmundson et al. 1995)—including research in the 15-mile reach—and they represent the best available science on flow needs of the endangered fish in this river segment. As such, we are no longer referencing Method 10. We have instead used the PBO recommendations to illustrate the status of flows as compared to the recommendations. There is one StateMod node within the 15-mile reach (09106150) and flow recommendations were specific to this same location, making it relatively easy to compare of gage and/or modeled streamflows with flows recommended for endangered fish.

Flow recommendations are expressed in the PBO (US FWS 1999) as (i) target peak day spring flows (PBO Table 1), (ii) recommended mean monthly flows for four different exceedance values, and (ii) volumes of water needed per 10-day period to achieve spring flow targets. It was beyond the scope of this effort to assess the status of current flow relative to this full set of recommendations. However, we did want to compare current conditions with some aspects of these recommendations with the intent of offering a relatively simple approach to quickly ascertaining how a given flow scenario performs relative to these recommendations. Specifically, do current conditions in the 15-mile reach meet recommended conditions? To assess this question, we focused our analysis on a subset of the recommendations, representing the highest and lowest flow conditions. Specifically, we focused our analysis on spring flow targets, mean flows for June flows (the month with the highest targets), and mean flows for August through October (the months with the lowest targets).

The “current” record we assessed was 20 years of gage data (WY1991-2010) from USGS 09106150 COLO RIVER BELOW GRAND VALLEY DIV NR PALISADE, CO. For our analysis, we compared actual exceedance values to those recommended in the PBO. The results of this analysis (Figure 2) demonstrate that the occurrence of flows exceeding the recommendations (US FWS 1999) were the exception, rather than the rule.

The construction of fish passes over several diversion dams has restored access between Palisade and Rifle as of 2008 (RIPRAP 2009). This then raises the question of flow response at other nodes, including De Beque Canyon (Cameo node 09095500) and the alluvial reach between De Beque and Rifle (De Beque node 09093700). Flow requirements for this Palisade to Rifle reach were investigated by Osmundson (2001). Final recommendations for spring flows were provided, and closely match those for the 15-mile reach because of similar bankfull flows. The author was limited to interim recommendations for summer flow due to data constraints, but these were also similar to the 15-mile reach. Achieving the flow recommendations for the 15-mile reach would, in most instances, exceed the flow recommendations for the Palisade to Rifle reach because of the large water diversions above Palisade. So, in order to simplify analysis and reporting, flow metrics are only reproduced here for the 15-mile reach with the expectation this will adequately represent flow constraints for pikeminnow in the upper Colorado basin. Readers interested in the realization of flow recommendations for the Palisade-Rifle reach are referred to Osmundson (2001) for detailed comparison of current flow conditions at Cameo to recommended flows for endangered fish (available at http://cwcbweblink.state.co.us/WebLink/DocView.aspx?id=133033&page=1&dbid=0).

Table 1 Target peak daily spring flows in the 15-mile reach (Palisade to Grand Junction) of the Colorado River. These are specified in (US FWS 1999) to support endangered Colorado pikeminnow and razorback sucker.

Peak flow (cfs)	Return period
>23,500	5 in 20 years
21,750	10 in 20 years
16,700	16 in 20 years
12,900	20 in 20 years

Table 2 From (US FWS 1999), "recommended mean monthly flows for the top of the 15-Mile Reach in cubic feet per second. Rate is the percent of years recommended for identified flows based on winter snowpack levels. For example, in the wettest 25 percent of years, flows in June should average at least 15,660 cfs; stated another way, this recommendation should be met in 5 of every 20 years. During low-water years, June flows should average no less than 6,850 cfs, and such a minimum should occur at a rate of no more than 4 in 20 years (20 percent)." To clarify, "Rate" is for a flow-interval (e.g. August flows should be between 810 and 1,240 cfs for no more than 20% of years) whereas "Exceedance" is cumulative (e.g. August flows should exceed 810 cfs for 100% of years). The rows we analyzed are in bold type.

Rate	25%	25%	30%	20%
Exceedence	25%	50%	80%	100%
JAN	1,630	1,630	1,630	1,240
FEB	1,630	1,630	1,630	1,240
MAR	1,630	1,630	1,630	1,240
APR	3,210	2,440	2,260	1,860
MAY	10,720	9,380	7,710	7,260
JUN	**15,660**	**14,250**	**11,350**	**6,850**
JUL	7,060	5,370	3,150	1,480
AUG	**1,630**	**1,630**	**1,240**	**810**
SEP	**1,630**	**1,630**	**1,240**	**810**
OCT	**1,630**	**1,630**	**1,240**	**810**
NOV	1,630	1,630	1,630	1,240
DEC	1,630	1,630	1,630	1,240

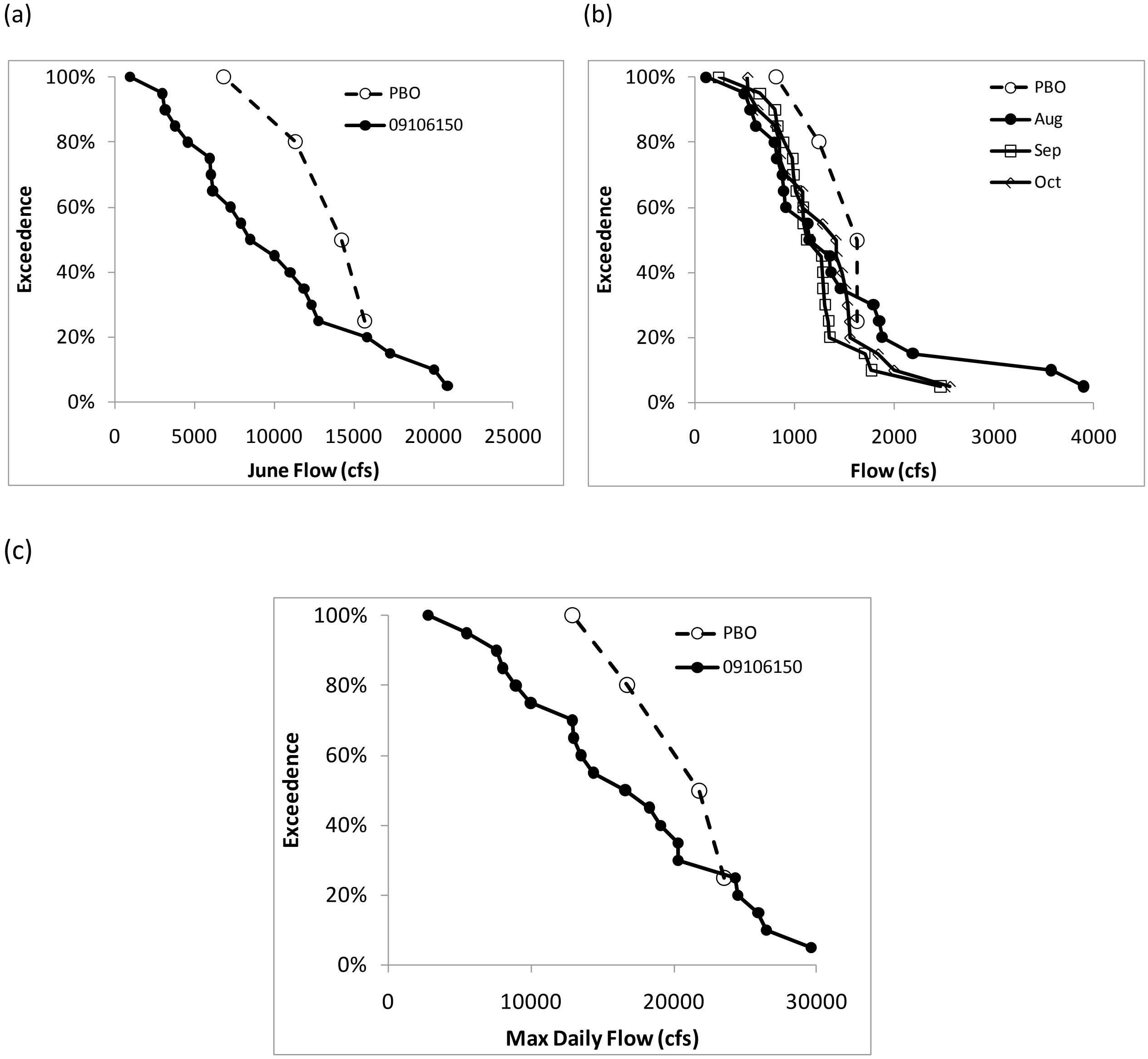

Figure 2. Exceedence of current flows (WY1991-2010) to flows recommended in the Programmatic Biological Opinion (PBO) for endangered fish at Palisade (US FWS 1999) using observed gage data: (a) mean flows for June, (b) mean flows for August, September, and October, and (c) annual maximum daily flows (Oct-Sep water year).

Non-listed warmwater fish

Wilding and Poff (2008) developed flow-ecology relationships for three other native warm-water species: roundtail chub, flannelmouth and bluehead sucker. All three species are the subject of a multi-state, rangewide conservation agreement (UDWR 2006). The roundtail chub is a Colorado state species of concern. The relationships developed in 2008—which approached each species separately—were revisited with the intent of develop a single relationship that could represent all three species (species-specific functions were presented in Wilding and Poff 2008 as Method 9). Firstly, it is worth reviewing the biology of these three species to be able to assess common responses to habitat conditions among the species. The following descriptions were adapted from Wilding and Poff (2008), with information added for this report on habitat and diurnal behavior.

*Biology of bluehead sucker (*Catostomus discobolus*)*

This species feeds on benthic algae and invertebrates. It is most commonly found in rocky riffle habitat (Ptacek et al. 2005), and has been observed at flow velocity of 0.4-1.4 m/s and depths of 0.3-1.5 m (Stewart and Anderson 2007, Bower et al. 2008). Daytime and nighttime habitats are similar (Beyers et al. 2001). The bluehead sucker matures at 4-6 years and >300 mm length in large rivers, 150 mm in small rivers, with some adults aged in excess of 20 years (Ptacek et al. 2005, Bower et al. 2008). Spawning benefits from high flow (Apr-July) and a low to moderate number of degree days (Muth and Nesler 1993).

Biology of flannelmouth sucker (Catostomus latipinnis)

This species feeds on benthic algae and invertebrates. It is a habitat generalist (Martinez et al. 2001, Rees et al. 2005a), and has been observed at flow velocity of 0.5-0.9 m/s and depths of 0.5-2 m (Stewart and Anderson 2007, Bower et al. 2008). Daytime and nighttime habitats are similar (Beyers et al. 2001, Rees and Miller 2001). The flannelmouth sucker matures at >400 mm length in large rivers, 200 mm in small rivers, and it can live for 30 years (Rees et al. 2005a, Bower et al. 2008). Spawning benefits from high flow (Apr-July) and a low to moderate number of degree days (Muth and Nesler 1993).

Biology of roundtail chub (Gila robusta)

This species feed opportunistically throughout the water column on plant matter, invertebrates and fish (Rees et al. 2005b). Daytime habitat of adults includes deep, low-velocity habitats with cover (Rees et al. 2005b), with increased use of shallow habitats at night (Beyers et al. 2001).

Comparing the three species, both suckers consume similar food but feed in different areas. Bluehead suckers are scrapers that are generally associated with cobble substrates, compared to flannelmouth suckers that feed on smaller substrates, including silt. Flannelmouth suckers are found in slower water than bluehead but, because greater depths are occupied by flannelmouth, the flows needed to produce suitable habitat may approach that of bluehead. For roundtail chub, their diurnal shift complicates direct comparison of habitat use. The diet and diurnal habitat use of adult roundtail chub implies active hunting behavior more like Colorado pikeminnow than the suckers. Roundtail chub were historically found at higher elevations than pikeminnow and lower elevations than cutthroat habitat.

Distribution

The distribution of the three species was reviewed in detail by (Bezzerides and Bestgen 2002) and the report, complete with distribution maps, is available at http://warnercnr.colostate.edu/larval-fish-lab-contributions/. The three species are confined to Western Slope streams and were commonly and recently collected from the mainstem Colorado River below Rifle. Historically, flannelmouth and roundtail were found farther upstream at least as far as Glenwood Springs, with bluehead sucker reaching Parshall (Bezzerides and Bestgen 2002). There are recent records of both flannelmouth and bluehead sucker further upstream than Rifle, including Dotsero on the Colorado River (Deacon and Mize 1997), the lower Eagle River (Woodling and Albeke 1999), and from the Roaring Fork below the Frying Pan confluence (Miller 2002).

Based on these distributions, we recommend limiting application of the flow-ecology methods for sucker in the upper Colorado basin to nodes below 7,000 feet elevation. More specific limits can be specified for the mainstem of the Colorado River - Radium at 6,850 ft (downstream of USGS 09058030). Likewise, on the Roaring Fork River a specific upstream limit at the Frying Pan confluence is recommended (6,590 ft). Suckers are likely present at higher elevations, but the cutoff is intended to constrain application of flow-ecology methods to sites where temperature is less likely an overriding constraint.

Flow-ecology methods for chubs and suckers

The same dataset used by Wilding and Poff (2008) was reanalyzed for this report. Anderson and Stewart (2007) gathered fish data across a wide range of flow conditions, representing gradients of flow modification, inter-year and site variability, using comparable methods. Sites included the Yampa, upper-Colorado, Gunnison and Dolores Rivers (see aerial photos Appendix 1). Mark-recapture raft electric fishing was carried out for all sites to estimate biomass per unit area (kg/ha). By employing data from rivers where temperature was not a major limiting factor, it was possible to distinguish the effects of flow. 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 in summer (daily average, U.S. Fish and Wildlife data).

Comparing the observed response to low flows, the three species are broadly similar (Figure 3), with generally increase % of maximum biomass as low flows increase. The two suckers are comparable, with the smaller bluehead sucker possibly benefiting more at lower flows (<500 ft^3/s). By comparison, Stewart and Anderson (2007) predicted bluehead to have higher flow requirements than flannelmouth. The response of roundtail chub to low flow was variable, with only a few site/years with >50% of maximum biomass. These few sites also fall short of the maximum biomass observed for the suckers (maximum 40 kg/ha for roundtail chub compared to 348 kg/ha for bluehead sucker and 180 kg/ha for flannelmouth sucker). There is not enough data to confidently claim the sucker relationship is representative of the flow response for roundtail chub. There are also important differences in feeding and habitat use by roundtail chub that might produce a divergent flow response. Subsequent analysis therefore excluded roundtail chub, instead using combined biomass of bluehead and flannelmouth suckers (% of maximum biomass) to provide a single response function representative of the suckers.

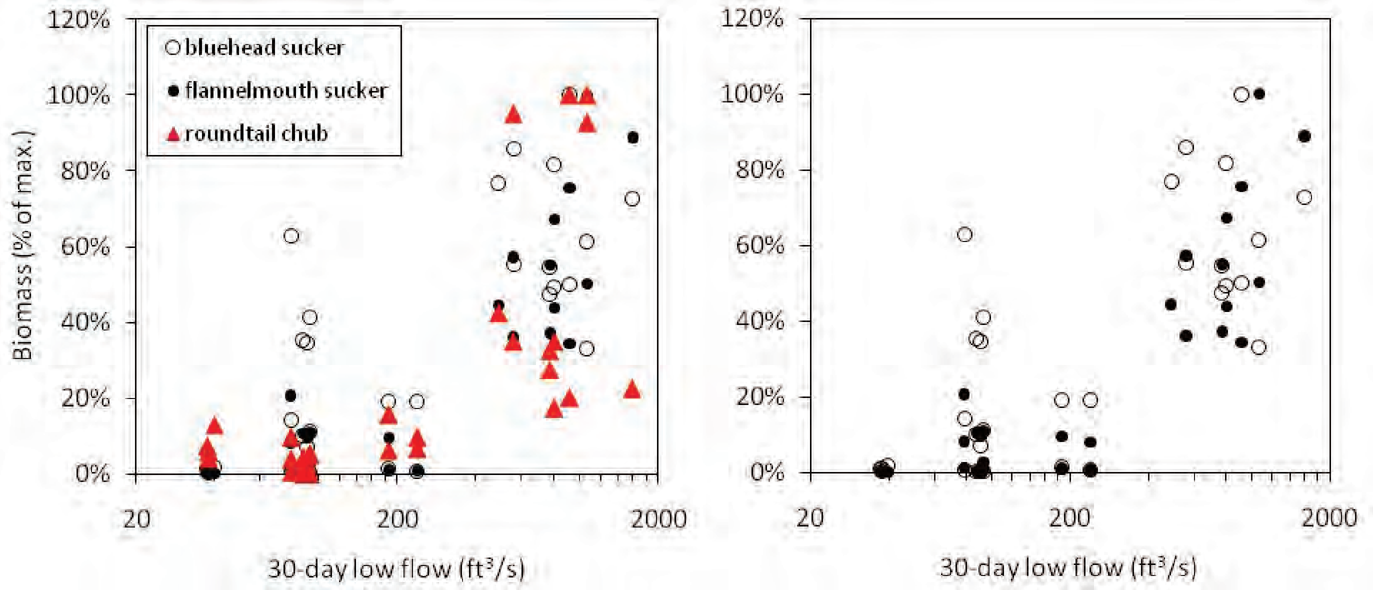

Figure 3 Comparing the response of the 3 fish species to low flow. Roundtail chub, bluehead and flannelmouth sucker are presented in left plot, with the right plot focusing on the two suckers for clarity. Data are sourced from Anderson and Stewart (2007). Fish biomass was measured in kilograms per hectare, and subsequently standardized by the observed maximum for this analysis. Low flow is quantified as the minimum 30-day moving average flow for July-November.

The WFET Pilot (CDM et al. 2009) used flow metrics provided by Anderson and Stewart (2007) and these were revisited using alternative flow metrics calculated from the same gages to determine which metric can best predict biomass response, with particular emphasis on metrics that are confidently calculated using StateMod model (based on daily time series). Using Indicators of Hydrologic Alteration (IHA; Richter et al. 1996) software, short-term flow minima (1, 3 and 7 day running mean) and extended minima (30 and 90 day) were calculated for the summer-autumn period (1 July to 30 November). It was necessary to isolate the low-flow season because low flows can occur at any time of year in the more regulated rivers and the summer-autumn minima also had the advantage of being better predictors compared to winter minima. Anderson and Stewart (2007) also used seasonal minima instead of annual minima presumably for the same reasons. The 7-day minima should be less sensitive to outliers than the 1-day minima used in the original WFET report, and gave a similar response (Figure 4). The 30-day minima produced a higher R^2 value than the pre-sampling 60-day average used by Anderson and Stewart (2007) (Figure 4). The use of a pre-sampling average flow is not directly applicable to future flow scenarios as we are not concerned with any specific date. All functions use Log_{10} transformed values of absolute flow, and the correlations were not improved using specific discharge (flow/watershed area).

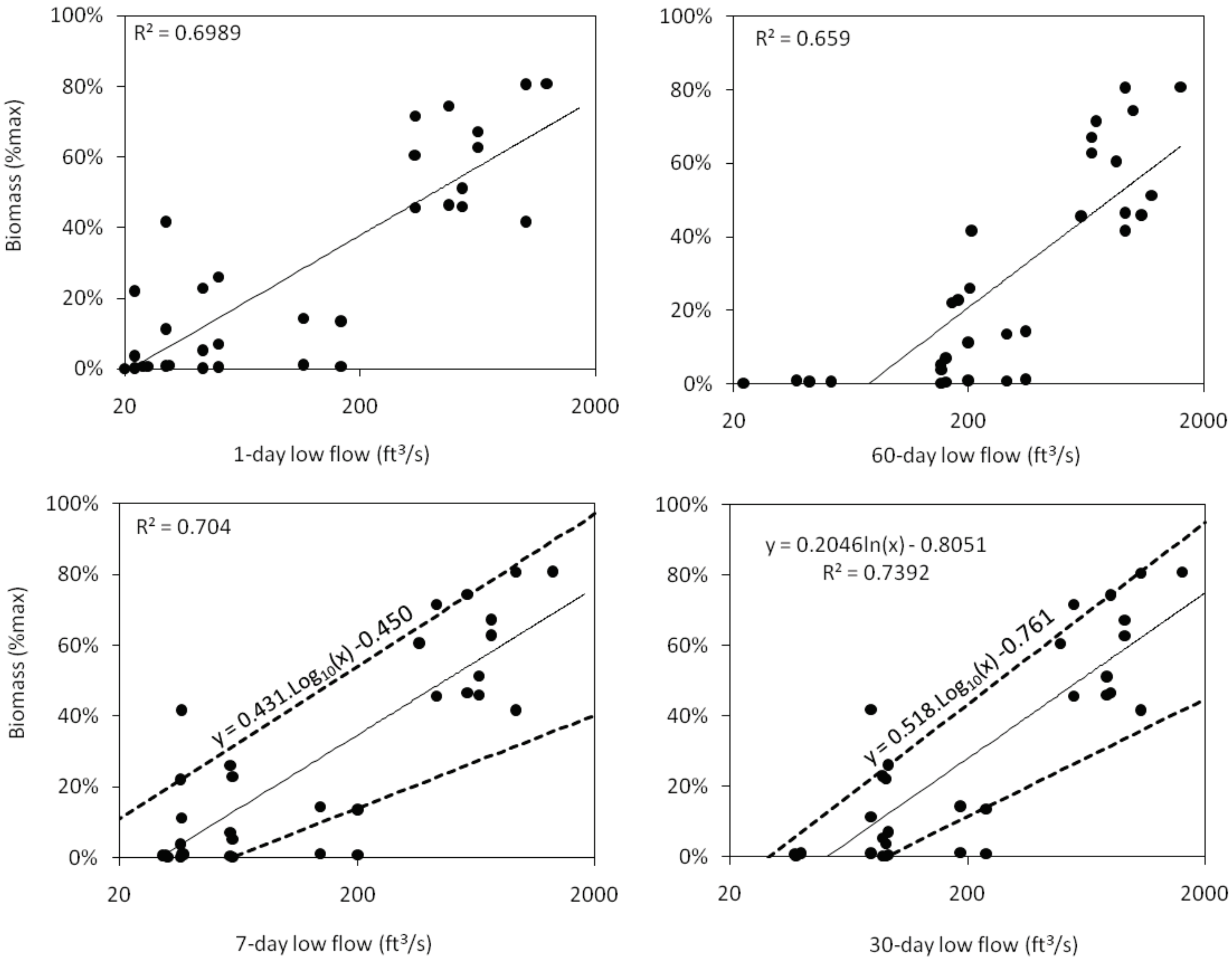

Figure 4 Comparison of response to four flow metrics by fish biomass (% of max. across all site/years) averaged for flannelmouth and bluehead sucker (each point is a site-year estimate). The top two plots use flow metrics sourced from Anderson and Stewart (2007) with the “60-day low flow” averaged over the 60 days prior to fish sampling (as per Wilding and Poff 2008). The lower two plots are the minimum running mean (7 and 30 day) during summer/autumn (1 July to 30 November). The R^2 values relate to the mean response (least squared regression), and quantiles are also fitted (using least absolute deviation) to the lower plots (10% and 90%ile, p<0.1 and <0.01 respectively) with equations given for the 90%ile (upper bound).

Although the above results (Figure 4) show a relationship between minimum flows and individual site-year values, it is more useful to consider biological response to long-term flow changes (e.g. contrasting 25 years with and without diversions). The long-term is of more interested because bluehead and flannelmouth suckers are long-lived fish (Ptacek et al. 2005, Rees et al. 2005a), and so the population observed any one year is a product of complex population dynamics over preceding years. We increased the temporal scale by averaging the annual monitoring data over a longer time step (in the absence of a population dynamic model for every reach in Colorado, we instead treat year to year variation as stochastic). The biomass data for each site were divided into two groups - a dry period (2002-2005) and a period of above average flows (1997-2001)[1], with flow averaging extending back an additional three years. The units of biomass were also changed from area based (kg/ha) to river-length based (kg/km) because standardizing by area (hectares in this case) factors out an important aspect of flow dependence - area increases with flow (temporally and spatially). This conversion was calculated using the flow for each sampling date and the relationships between surface area and flow (derived using Table II-4 from Anderson and Stewart 2007).

These refinements clarified the flow response for suckers. Considering only the temporal component, the dry-period biomass of suckers was less than the wetter-period biomass for all sites (paired t-test p-value 0.03 performed on averages, except the Gunnison which lacked pre-drought biomass data). Sucker biomass increased with flow both over time and between sites (Figure 5), though there remains some scatter about the mean response function (left plot). Researchers report higher numbers of sucker in rocky areas that provide stable substrate for algae and other food (e.g., Ryden 2001). This explains some of the variability in the flow response observed here. Specifically, the residuals from the mean flow response are positively correlated with channel slope ($R^2 = 0.54$, see footnote[2]), which is expected since more cobble riffles—which is better sucker habitat—typically occur in steeper reaches.

An additional source of variability is the suppression of sucker biomass by high densities of introduced fish. For example, the 2002 drought was associated with a dramatic increase in numbers of smallmouth bass in the Yampa River (Anderson and Stewart 2007, Bestgen et al.

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[1] For example, 30 day low flows for Yampa at Maybell averaged 208 ft^3/s for 1997-2001 compared to 76 ft^3/s for 2002-2005. Annual series statistics for the full record (1917-2009) include a 30%ile low flow of 113 ft^3/s and 70%ile of 226 ft^3/s.

[2] Multiple linear regression model: %max biomass (kg/km) = 0.2265.Ln(30 day min cfs) + 268.8.slope - 1.208. For N = 14, F-stat = 70.76, regression P-value < 0.001, Adjusted $R^2 = 0.915$. In this model, the P-value for the slope coefficient (268.8) was 0.001. Model would only apply to warmwater streams, as temperature is not included.

2007). Likewise, introduced fish may pose an important constraint on sucker biomass in the Dolores River (Anderson 2010). A second response function was therefore developed for the flow period dataset, which omits data points where substrate stability or introduced fish may be primary constraints on sucker biomass (dashed line, Figure 5). The slope threshold used for habitat suitability (sites excluded if slope <0.10%) aligns with the geomorphic classification used for the WFET (Bledsoe and Carlson 2010), distinguishing moderate-energy reaches from low-energy. The recommended flow-ecology method for sucker is:

% maximum native sucker biomass = 0.1026 x 30-day low flow$^{0.3021}$

This revised function acknowledges that factors in addition to flow may also constrain native fish populations in the Dolores and Yampa Rivers. Eliminating data collected where non-native fish are present also presumes that flow was not a primary mechanism for the impact of introduced fish.

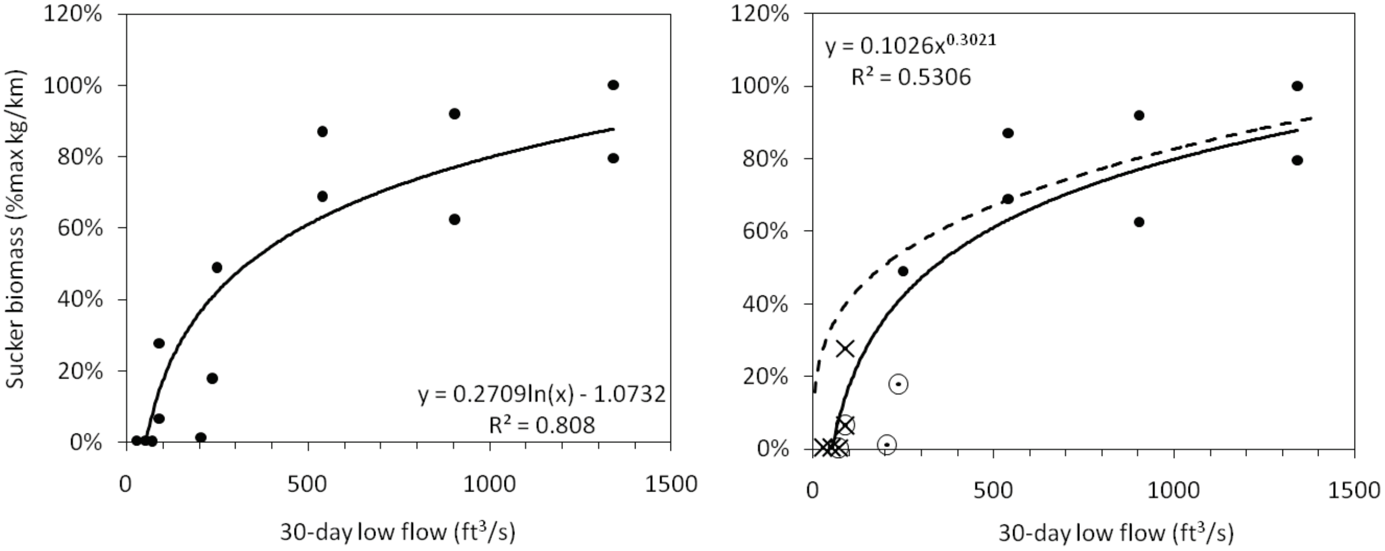

Figure 5 The response of sucker biomass to flow, increasing the temporal scale to flow periods (from years in previous plots). Each site is represented by just two points (average biomass pre- and post-2002 drought). The biomass units were changed from area based (kg/ha) to river length based (kg/km) to isolate biomass from changes in width with flow. The right plot differs from the left by circled data points for low energy streams (slope <0.10%) and crossed points where biomass may be suppressed by introduced fish (Yampa post 2002 and both Dolores periods). The dashed regression line describes the mean response for only steeper streams less impacted by introduced fish (ANOVA p-value <0.01 for both regression lines).

Sucker method validation

Validation of the sucker method employed two datasets – the first a spatial dataset (multiple sites sampled once), and the second a temporal dataset (repeat annual monitoring).

The first dataset describes abundance of suckers at 15 sites in the upper-Colorado basin (Deacon and Mize 1997). The flow metric (30 day minimum flow for July-November) was calculated from relevant StateMod nodes (for 4 sites) and USGS stream gages (for 11 sites). In three cases, the gage record did not cover the fish monitoring period, and so an extended record was synthesized from more distant gages with overlapping records. Stream temperature was reported by (Deacon and Mize 1997) and is used here as an alternative predictor to flow (they did not specify the duration of temperature monitoring). Abundance was measured as the number of fish caught per site, summed across bluehead and flannelmouth suckers.

The cold water streams draining the Rocky Mountains are expected to test the lower-thermal limits of suckers. Water temperature increased with flow ($R^2 = 0.32$) – the exceptions being four small streams at low elevations. The correlation clearly works in favor of sucker abundance (Figure 6) with higher abundance in warmer larger streams. Both variables are important and neither variable is adequate on its own to explain differences in abundance between sites. Residuals from the temperature relationship were still positively correlated with flow (Figure 7). Removing the temperature effect also produces a flatter response to flow that better matches the sucker method (Figure 7).

The sucker abundance metric (total number of suckers caught) is influenced by fishing effort which was not equal across sites (wadeable sites were electric fished over a 450-650 ft reach and nonwadeable sites boat-electric fished over a 1,500-3,000 ft reach). Site specific effort was not described by the authors, but presumably the five largest rivers were boat electric fished (these do not appear wadeable from aerial photos). The shortcomings of the abundance metric prevents development of a predictive model from this dataset, but adds weight of evidence in validating the flow-ecology relationship derived in this report using an independent dataset.

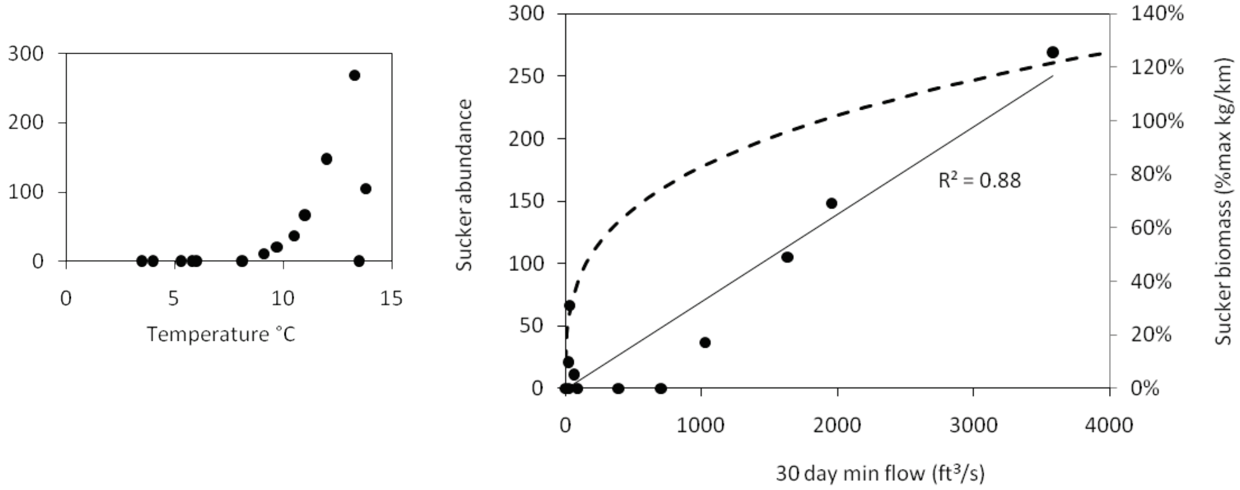

Figure 6. Association between suckers (bluehead plus flannelmouth) and flow for sites monitored by USEPA in the upper Colorado basin (Deacon and Mize 1997). The flow metric is the 30-day minimum for July-November, averaged over the sampling year and five-years prior. Abundance is the number of suckers caught. The dashed line is the sucker method from Figure 5 overlaid for comparison, with units on the right axis (%maximum biomass). Temperature is also plotted to the left as a co-determinant of sucker abundance.

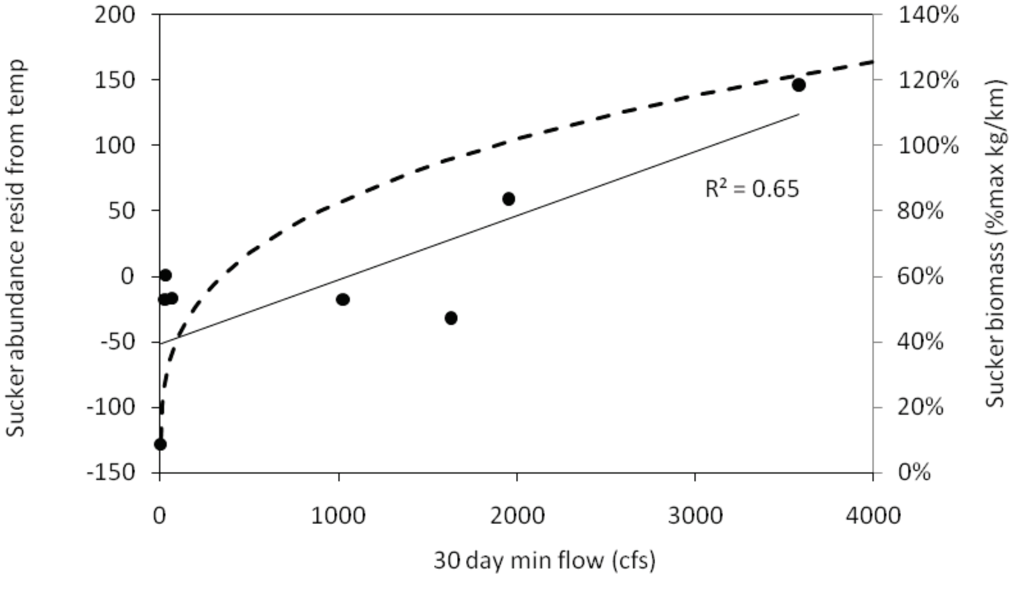

Figure 7. Residual flow response of sucker abundance after removing the effect of temperature. Only sites below 7,000 feet are presented (zero abundance at sites above this elevation). The dashed line is the sucker method from Figure 6 overlaid for comparison with units on the right axis (%maximum biomass kg/km). Otherwise as per Figure 6.

The second validation dataset describes changes over time for the sucker population of the San Juan River (Ryden 2010). The SJRIP (San Juan River Basin Recovery Implementation Program, www.fws.gov/southwest/sjrip) monitors fish populations annually for the mainstem San Juan River between the Animas River confluence and the Colorado River confluence (180 river miles). The suckers (bluehead in particular) are more abundant in reaches of the San Juan with more cobble substrate, notably Reach 6 below Farmington (channel slope 0.2%) (Ryden 2001). These stony reaches are assumed to drive interannual dynamics, despite using data from the extended monitoring area (180 river miles). Raft electric fishing data was analyzed by (Ryden 2010) to estimate CPUE (catch per unit effort - fish caught per hour) for the various species and size classes. For our validation analysis, CPUE data were extracted from the (Ryden 2010) report. Data for adult suckers (bluehead plus flannelmouth) were used in an effort to provide a better correlate of biomass than juveniles (the sucker method was developed from biomass data).

The number of adult sucker caught was not correlated with low flow (Figure 8). Populations of adult sucker were relatively stable over the monitoring period (1999-2009). Flows varied during this period, but in every year remained about 200 cfs. Changing from an annual time-step to flow periods that are more analogous to the flow periods used in the recommended sucker method also illustrates the stability of sucker populations in the San Juan. Using longer flow periods also illustrates that long-term low flows on the San Juan were also relatively stable during from 1999-2009. The monitoring results are therefore consistent with the sucker method which predicts little change in sucker populations in the absence of long-term reductions in flow, particularly at higher levels of flow (see dashed line in Figure 8).

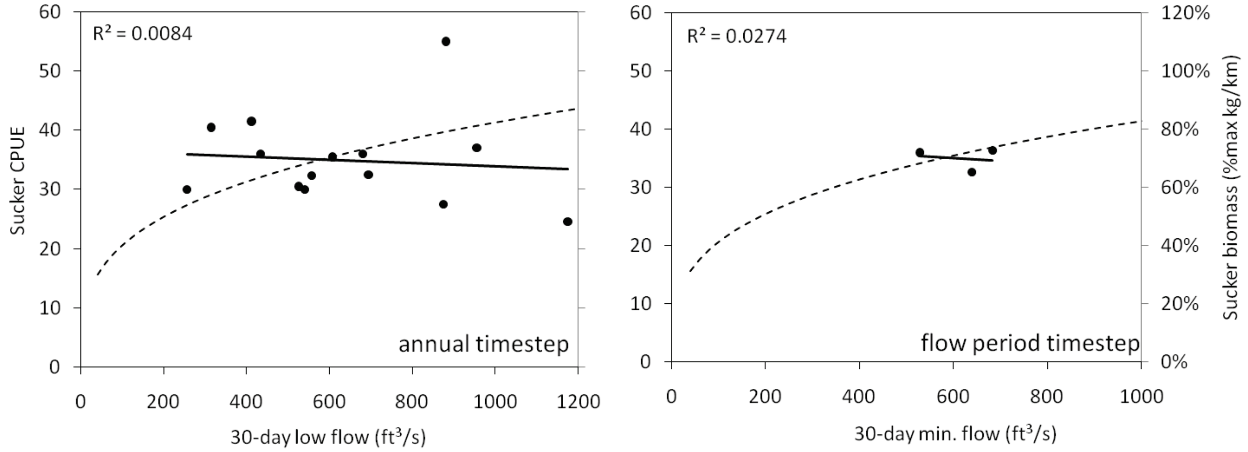

Figure 8

Correlation between number of suckers (bluehead plus flannelmouth) and flow for 180 miles of the San Juan River for the period 1996 to 2009 (data from Ryden (2010) and Ryden (2003)). The flow metric is the 30-day minimum for July-November at USGS 09371010 (Four Corners). The number of adult suckers (flannelmouth > 409 mm, bluehead >299 mm) caught per hour provides a standardized CPUE (catch per unit effort). The left plot presents annual monitoring results individually, and the right plot uses a longer time-step to represent flow periods (average 1996-01, 2002-04, 2005-09). The sucker method relationship is also presented (from Figure 6) as a dashed line using different units on the right y-axis (%maximum biomass).

Risk Classes for the Sucker Method

The sucker method can be used to contrast natural and altered flows, or other flow scenarios. Risk classes are recommended following input from fish experts based on the expected change in % maximum sucker biomass, as follows: low risk = 0-10% reduction in maximum biomass; minimal risk = 10 - 25% reduction; moderate risk 25-50% reduction; high risk = 50-100% reduction.

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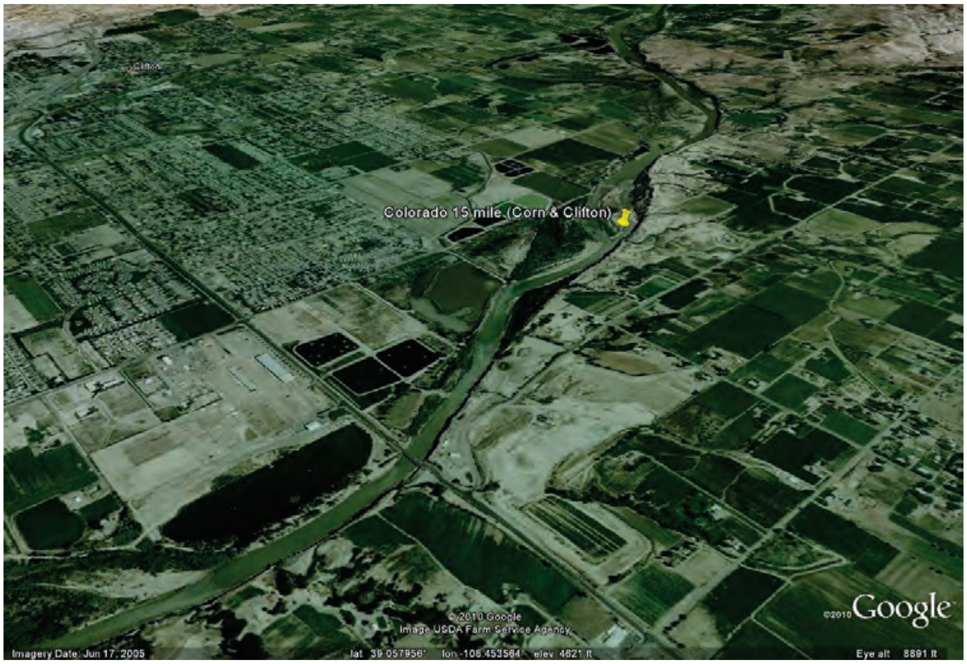
Appendix 1 – Site Photos

Aerial photos from Google Earth of approximate locations fished by Anderson & Stewart (2007). This is intended to depict general reach morphology rather than precise fishing locations.




Yampa at Duffy
Image © 2011 DigitalGlobe
© 2010 Google
Google
Imagery Date: Sep 29, 2005
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Eye alt 11320 ft


Yampa at sevens
Image USDA Farm Service Agency
© 2010 Google
©2010 Google
Imagery Date: Jun 17, 2005
lat 40.498911° lon -108.308419° elev 5831 ft
Eye alt 10418 ft


Yampa at Lilly
Image USDA Farm Service Agency
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©2010 Google
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lat 40.457293° lon -108.415092° elev 5826 ft
Eye alt 9950 ft


Gunnison at Delta
Image © 2011 DigitalGlobe
© 2010 Google
©2010 Google
Imagery Date: Mar 23, 2006
lat 38.742778° lon -108.123351° elev 4899 ft
Eye alt 10070 ft


Gunnison at Escalante
Image © 2011 DigitalGlobe
© 2010 Google
Image USDA Farm Service Agency
©2010 Google
Imagery Dates: Jun 17, 2005 - Oct 17, 2005
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Eye alt 11012 ft

Appendix F

Riparian Vegetation Methods for the Watershed Flow Evaluation Tool

RIPARIAN VEGETATION METHODS FOR THE WATERSHED FLOW EVALUATION TOOL

A report to the Non-Consumptive Needs Committee of the Colorado Basin Roundtable

December 2010

Thomas K. Wilding Colorado State University

John Sanderson The Nature Conservancy

EXECUTIVE SUMMARY

Riparian vegetation is a key element of riverine ecosystems, providing many ecological, aesthetic and economic benefits, including terrestrial wildlife habitat structure, food resources, stabilizing geomorphic properties along banks and floodplains, and energy subsidies to aquatic and terrestrial ecosystems (Pusey and Arthington 2003). Riparian vegetation composition, structure and abundance are governed to a large degree by river flow regime and flow-mediated fluvial processes (Merritt et al. 2009). Streamflow regime exerts selective pressures on riparian vegetation, resulting in adaptations to specific flow attributes (Merritt et al. 2009), and riverine species have evolved life history strategies primarily in direct response to natural flow patterns (Bunn and Arthington 2002). Widespread modification of flow regimes by humans has resulted in extensive alteration of riparian vegetation communities (Merritt et al. 2009). Altered flow regimes may cause changes in plant species richness (Jansson et al. 2000, Nilsson and Svedmark 2002), plant growth and productivity (Stromberg & Patten, 1990), community composition (Merritt & Cooper, 2000; Merritt & Wohl, 2006) and loss of riparian forests (Rood & Mahoney, 1990; Braatne et al., 2007).

The Roaring Fork Pilot WFET (Watershed Flow Evaluation Tool) developed a quantitative relationships between flow alteration and riparian vegetation using many literature sources (Wilding and Poff 2008). The source literature covered a diverse range of vegetation types, including cottonwood, willow and herbaceous plants. In response to feedback received on the pilot as well as peer-review comments received during and after an expert workshop, this report refines the approach and narrows the application of the flow-riparian relationship. Specific changes and refinements to the methods used in the Roaring Fork pilot include:

1) Flow-ecology relationships are now described for three riparian types: i) cottonwoods on low- and moderate-gradient, meandering (open, or unconfined) rivers, ii) cottonwoods in moderate-gradient rivers of confined valleys and high-gradient rivers in unconfined valleys, and iii) willows in low-gradient, unconfined valleys.
2) Quantitative flow-ecology relationships were developed only for the two cottonwood types. Despite some evidence of willow dependence on floods (Cooper et al. 2006), we lacked

sufficient data to quantify this dependence over a range of flow alteration. For willows, the flow ecology relationship is described only conceptually.

3) Flow-ecology relationships are now applied only in the specific elevation ranges and select geomorphic settings where that relationship is expected to exist.
4) A new, large data set on cottonwoods (Merritt and Poff 2010) allowed for development of a robust quantitative flow-ecology relationship for cottonwoods in low-gradient, unconfined geomorphic settings.
5) Flood magnitude alteration is calculated only in the 30% of years with the highest mean annual flow.
6) No hydrographs are developed based on break-points between risk classes, in contrast to the Roaring Fork pilot.

RIPARIAN FLOW-ECOLOGY CURVES RECOMMENDED FOR APPLICATION IN THE COLORADO WFET

Cottonwood in Unconfined (wide valley) settings

Geomorphic setting where applied: Moderate-energy unconfined, Low-energy floodplain, and Glacial trough. *Elevation where applied*: <9600 feet

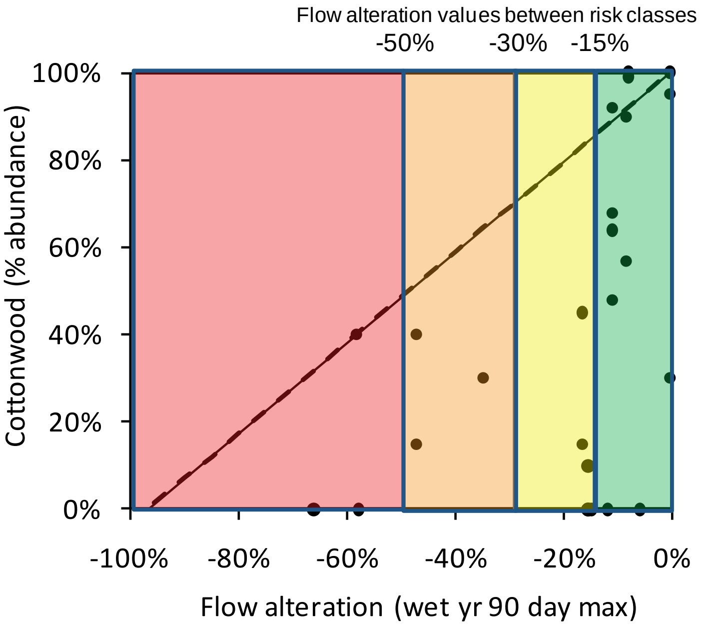
Two quantitative flow-ecology relationships exist for cottonwood in unconfined settings, one for adult cottonwood abundance and the other for cottonwood recruitment.

Adult cottonwood – The hydrologic metric for adult cottonwood is the change in average 90-day maximum flow in wet years only between current and undeveloped scenarios. “Wet years” are those in the top 30th percentile for mean annual flow in the undeveloped flow time series. Cottonwood abundance is calculated as:

- If flow alteration is >0% (i.e. flow augmentation) then cottonwood abundance = 100%
- If flow alteration is ≤0% then %abundance = 1.038 x %flow alteration + 1.005.

Risk classes:

Risk Class	Flow alteration	Justification for change to next higher risk class
Low	0 to -15%	Natural break in data—beyond flow alteration of -15%, no abundance greater than approximately 45%.
Moderate	-15% to -30%	Twice the risk of ‘low’.
High	-30% to -50%	Natural break in data—only one non-zero value at flow alteration beyond ->50%
Very High	-50% to -100%	No data beyond flow alteration of more than -70%.



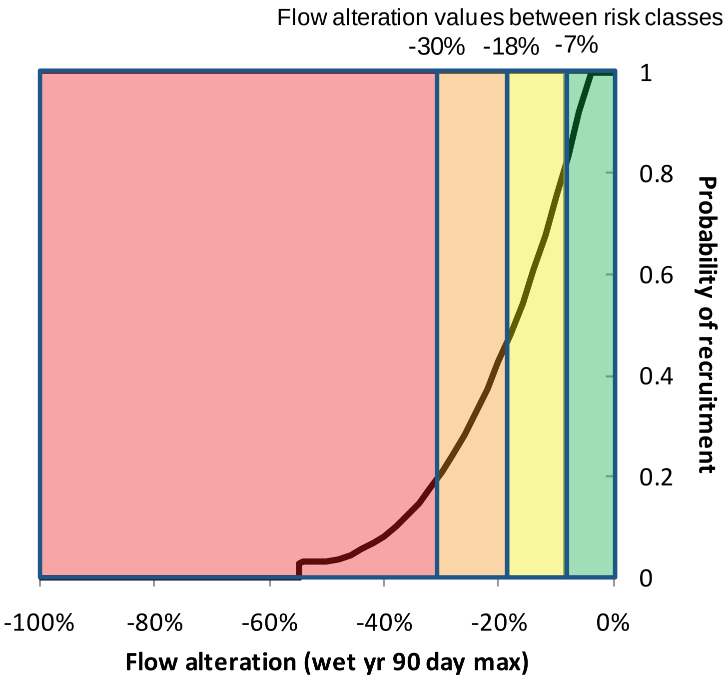
Flow-ecology relationship and risk classes for adult cottonwood in low- and moderate-gradient, unconfined settings.

Cottonwood recruitment – The hydrologic metric is the same as for adult cottonwood and is also calculated for only wet years. The probability of cottonwood recruitment is calculated as:

- If flow alteration is 0% to -4% then recruitment = 1.
- If flow alteration is -4% to -55% then recruitment = 2.91 x %flow alteration^3 + 7.27 x %flow alteration^2 + 5.26 x %flow alteration + 1.21.
- If flow alteration -55% to -100% then recruitment = 0.

Risk classes:

Risk Class	Flow alteration	Justification for change to next higher risk class
Low	0 to -7%	At flow alteration of -7%, probability of recruitment is reduced to 0.9.
Moderate	-7% to -18%	At flow alteration of -18%, probability of recruitment is reduced to 0.5.
High	-18% to -30%	At flow alteration of -30%, probability of recruitment is reduced to 0.2.
Very High	-30% to -100%	At flow alteration of -30% to -55%, probability of recruitment is less than 0.2.



Flow-ecology relationship and risk classes for cottonwood recruitment in low- and moderate-gradient, unconfined settings.

Cottonwood in Confined settings

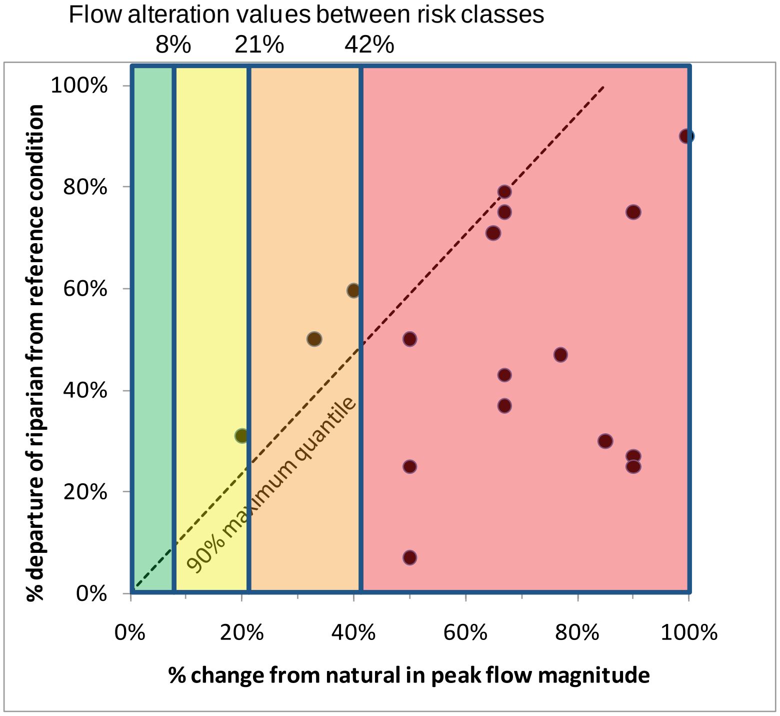
Geomorphic setting where applied: Moderate-energy confined. *Elevation where applied*: <9600 feet

% Departure of riparian from reference conditions: Calculated using Method 7 from Wilding & Poff (2008). Unlike the previous two cottonwood metrics, this metric is calculated using data from all year types. The hydrologic metric is calculated as:

$$\%\text{ departure from reference condition } = \frac{\text{Annual Peak Daily Flow}_{\text{current}} - \text{Annual Peak Daily Flow}_{\text{baseline}}}{\text{Annual Peak Daily Flow}_{\text{baseline}}}$$

Risk classes:

Risk Class	Flow alteration	Justification for change to next higher risk class
Low	0 to 8%	At flow alteration of 8%, expected departure from reference condition is 10%.
Moderate	8% to 21%	At flow alteration of 21%, expected departure from reference condition is 25%. Maximum measured departure in this range is 31%.
High	21% to 42%	At flow alteration of 42%, expected departure from reference condition is 50%.
Very High	42% to 100%	In this range, measured departure from reference is at least 20% and as high as 90%.



Flow-ecology relationship for cottonwood in moderate-gradient confined settings and high-gradient unconfined settings.

Willow in Unconfined settings

Geomorphic setting where applied: Moderate-energy unconfined, low-energy floodplain, Glacial Trough. *Elevation where applied*: >8000 feet

Willow shrubland:
Evaluate %alteration of peak-flow (annual 1-day maximum or wet year 30-day maximum). We do not have data that describes the manner in which willow shrublands change as flow changes. Importantly, it is possible that beaver mitigate the negative impacts of reduced peak flow. See the Willow section of this report for discussion of willow response to flow alteration and hypothesized models.

Risk Classes: Due to the conceptual nature of this flow-ecology relationship, no risk classes are recommended.

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INTRODUCTION

The riparian zone is the area adjacent to a stream, and is distinguished by the influence of flood disturbance and more water in general than surrounding land. It represents a critical area for wildlife, including those inhabiting surrounding land and the stream itself (for an introduction see Gregory et al. 1991, Naiman and Décamps 1997, Patten 1998). The focus here is on riparian vegetation, which, among other things, provides critical habitat for terrestrial species (for example, game species and neotropical migrants), provides the carbon and energy that supports aquatic food webs, plays essential roles in supporting streambank and in-channel habitats, and has tremendous aesthetic value (Pusey and Arthington 2003).

The Watershed Flow Evaluation Tool (WFET) describes relationships between flow and river species and ecosystems. Wilding and Poff (2008) used many literature sources to develop a single quantitative relationship between flow alteration and riparian vegetation (Wilding and Poff 2008). The source literature used by Wilding and Poff (2008) covered a diverse range of vegetation types, including cottonwood, willow and herbaceous plants. This current report describes the development of additional methods--focusing on cottonwood forest and willow shrubland—that are described using recently published data, applied to specific geomorphic settings, and that have been subjected to additional peer review.

Flow ecology relationships are described quantitatively where sufficient data allowed reliable modeling of the relationship and qualitatively or conceptually in other cases. It is important to recognize that the complexity of river ecosystems precludes modeling all aspects of the system. While quantitative riparian flow-ecology relationships are available only for cottonwood, basic ecological principles suggest that the flow regime necessary to sustain cottonwood and willow is also expected to sustain the physical biological processes that support the broader riparian ecosystem, including processes of disturbance, nutrient cycling, and water flows. Cottonwood are therefore offered as an indicator of flow adequacy for riparian ecosystem as they are pervasive in the Colorado River basin and good data exist to describe the flow-ecology relationship.

The mechanisms by which establishment and growth of cottonwoods depend on flow are well established (Friedman et al. 1995, Scott et al. 1996, Auble and Scott 1998, Mahoney and Rood 1998, Cooper et al. 1999, Karrenberg et al. 2002, Shafroth et al. 2002, Rood et al. 2007, Stromberg et al. 2007). Recruitment from seed in wide valleys is particularly well understood. Floods create bare surfaces (from erosion or deposition) and remove competing plants, providing moist, sandy and unshaded conditions for seed germination. In semi-arid areas, flow recession must be gradual enough for the roots of seedlings to keep pace with dropping water levels (less critical in humid regions). The magnitude, frequency and timing of flows (within and between years) all come into play for a successful recruitment event. The right flow conditions are therefore required for seedling growth, but are not necessarily sufficient for survival to the age of reproducing adults. It may be three years before the roots of seedlings achieve reliable access to groundwater, assuming they are not eaten, burned or washed away (Auble and Scott 1998, Cooper et al. 1999, Polzin and Rood 2006, Rood et al. 2007). Asexual recruitment (i.e. suckering) has also been described in flow-related mechanistic terms (Roberts 1999, Polzin and Rood 2006). Cottonwood survival and growth depends on base flows in addition to flood flows (Stromberg and Patten 1991), but the base flow relationship is not described in this report because we lack sufficient data to develop a generalized relationship between base flows and cottonwood health

GEOMORPHIC SETTING IS MORE IMPORTANT THAN SPECIES

To understand the role of species, reproductive traits and geomorphic setting in determining flow dependence of cottonwood and willow, experts were invited to attend a Riparian Workshop and provided valuable input as well as direction for the literature review. Within Colorado there are several species of *Populus* that depend on the river to varying degrees (all species except aspen - *Populus tremuloides*). The sub-genus Section classification of *Populus* is more useful than species level classification, in the context of this report, as it better distinguishes the reproductive strategies of *Populus*. The section *Aegiros* (broadleaf cottonwoods) includes subspecies of *P. deltoides* (subspecies *monilifera*, commonly known as Rio Grande cottonwood and plains cottonwood) and *P. fremontii* (subspecies *wislizenii*, commonly known as Fremont cottonwood). These grow at lower elevations in Colorado (<6500 ft) and reproduce primarily from seed (Rood et al. 2007).

The other *Populus* section, *Tacamahaca*, is represented in Colorado by *Populus angustifolia,* commonly known as narrowleaf cottonwood. Literature for black cottonwood (*Populus trichocarpa*[1]) was also reviewed as this *Tacamahaca* section species helps us understand the transition of cottonwood traits (particularly fluvial reproductive traits) in response to geomorphic (valley shape) and temperature gradients between semi-arid plains and high mountains (Gom and Rood 1999). Narrowleaf cottonwood are found at higher elevations (5,200-9,600 ft; Carsey et al. 2003) than broadleaf cottonwoods, with *angustifolia-deltoides* hybrids (*P. x acuminate*, Eckenwalder 1984) occasionally abundant at overlapping elevations (5200-6,500 ft; Carsey et al. 2003).

Asexual reproduction is often dominant or co-dominant for narrowleaf cottonwood, in contrast to *Aegiros* cottonwood that rely on sexual reproduction through seed dispersal (Rood et al. 1994, Rood et al. 2007). Asexual reproduction in narrowleaf is predominantly through root-suckering, where injury can trigger “new” trees to grow from the roots of existing adults (rather than from broken or abscised branches, Rood et al. 2003). Root-suckering can be triggered by

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[1] Also referred to as *Populus balsamifera* subsp. *trichocarpa*.

floods (Polzin and Rood 2006), and so can resemble sexual reproduction in *Aegiros* cottonwood. Root-suckering is expected to be a more effective reproductive strategy where the growing season is short (high elevations) and channel forming floods are less frequent (Patten 1998, Rood et al. 2007). Other disturbances can trigger asexual reproduction (colluvial movement on coupled slope slides, fire, herbivory; Rood et al. 2007), and this may negate consideration of flow alteration for recruitment of narrowleaf cottonwood in highly-coupled steep streams (narrow valleys and canyons), (Samuelson and Rood 2004).

Narrowleaf cottonwood is similar to broadleaf cottonwood in many respects, but successful recruitment is often associated with larger flood events (5-15 yr, compared to 2-5 yr events for *P. deltoides*). This distinction in the flow response appears to be a consequence of climatic and geomorphic gradients, which dictate a shift in reproductive strategies at higher elevations. Baker (1990) estimated good “seedling years” for narrowleaf cottonwood every 3.4 years on average, but “stand-origin years” for adult trees were less frequent at 10-15 years (true seedlings were not distinguished from root suckers). This study was completed on a confined section of the Animas River downstream of Silverton[2]. More frequent floods (e.g. 3 yr return) facilitate seedling germination, and this is probably sufficient for recruitment in wide valley settings where meandering can carry the river away from last year’s seedlings (Rood et al. 2007). But, in the steeper, more confined rivers where narrowleaf cottonwood often occur, meandering is confined so the river is more likely to scour last year’s seedlings. Bigger floods are therefore required to create bare colonization sites that are high enough above the frequently disturbed channel (Auble and Scott 1998, Polzin and Rood 2006). The coarser bed material in steep, confined valleys (>>2% slope, valley width <7x bankfull width) also necessitates a larger flood event to initiate bed movement (Ryan 1997). Growing seasons are short at higher elevation, further reducing the success rate of seedlings because of slow growth (Kalischuk et al. 2001). Seedling reproduction is therefore a riskier strategy in this setting, raising the importance of root-suckering for stand survival (Rood et al. 2007). Polzin and Rood (2006) suggested flow recession and low flows are less important for successful recruitment in the northern Rocky Mountains compared semi-arid areas farther south, because river flow is less likely to constrain seedling survival in cool moist environments.

Both sections of cottonwood depend on flow in a similar manner in wide valleys, where rivers are free to meander, shift and change (Patten 1998). In this setting we find the largest cottonwood forests and also the most flow-dependent forests (Gregory et al. 1991, Scott et al. 1996, Willms et al. 2006, Rood et al. 2007). Snowmelt is critical for cottonwood in this setting, with floods recurring every 3-5 years that provide the right conditions for germination and survival (Scott et al. 1996, Rood et al. 2007). This appears to hold true for root-suckering

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[2] Study completed in the Animas Canyon. This has a 2% slope for the 10 km river section between 2390 and 2575 m elevation, and colluvial deposits are visible reaching the channel viewed from aerial photos in Google Earth.

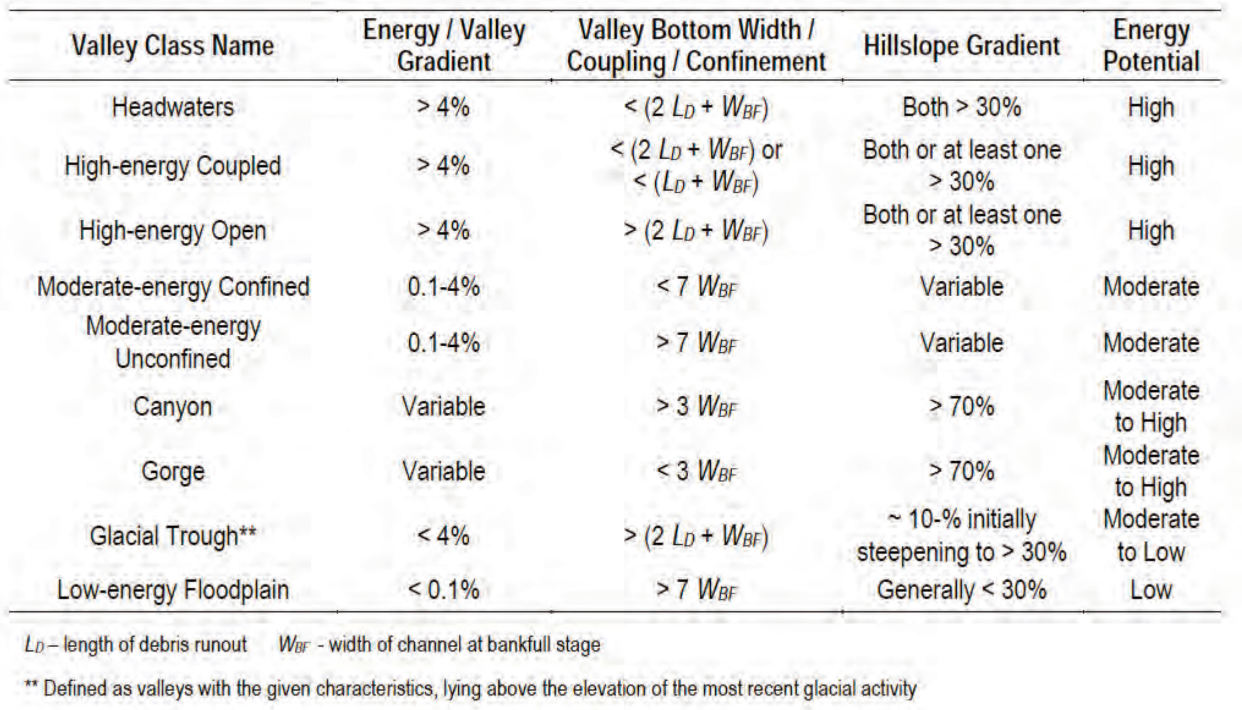
species, as demonstrated for narrowleaf cottonwood forests on the Yampa River by Richter and Richter (2000), and seedling recruitment of black cottonwood in “parkland” reaches by Samuelson and Rood (2004).

The geomorphic and climatic differences do not simply discriminate where each species is found, they are directly responsible for the relationship between flow and cottonwood recruitment. Therefore geomorphic classification is a better indicator of flow dependence than the species or section of cottonwood. So rather than applying the flow-ecology method to a given species, we should instead develop methods that are specific to the geomorphic settings that favor fluvial dependence as a reproductive strategy in riparian cottonwood (Merritt et al. 2009). This approach is further supported by the converse situation where *P. deltoides* recruitment is associated with infrequent flood events (10 yr return) in confined valleys (Auble and Scott 1998). Therefore wide valleys with low slopes are more likely to support cottonwood stands that depend on flow for successful recruitment. The greater fluvial dependence also increases the importance of flow management for riparian health in this setting.

DEFINING THE GEOMORPHIC SETTING

The relationship of flow to riparian vegetation is best considered within a geomorphic context, as it is the valley landform that determines the occurrence of riparian vegetation (type and extent) and their response to change in flow over time (Gregory et al. 1991, Scott et al. 1996, Rood et al. 2007). Methods were therefore developed to classify reach geomorphology in the Colorado basin. In a parallel investigation, Bledsoe and Carlson (2010) developed a geomorphic classification system for Colorado streams at the reach scale (Table 1). Processes under consideration here include valley confinement, where unconfined valleys (>7 x the bankfull channel width) allow streams to reach a sinuosity >1.5 and produce a wider flood zone with lower water velocities – conditions conducive to developing extensive riparian vegetation. Groundwater tables in wide valleys are more dependent on stream flow, and therefore flow is more important for riparian vegetation in this setting (cf. confined - Dawson and Ehleringer 1991). Valleys strongly coupled with adjacent hillslopes (where the valley width is less than 2x the bankfull channel width) are narrow enough for slides and rockfalls to reach the stream channel, potentially overwhelming the effect of stream processes on riparian vegetation, especially given the narrow zone of flood influence. Valley slope is an important determinant of stream power, and therefore processes creating riparian habitat such as sediment transport (erosion and deposition). Low valley slopes are required for developing sinuosity and are often associated with wide valleys.

Table 1. Geomorphic classification of Colorado streams from Bledsoe and Carlson (2010).



Valley Class Name	Energy / Valley Gradient	Valley Bottom Width / Coupling / Confinement	Hillslope Gradient	Energy Potential
Headwaters	> 4%	$< (2\,L_D + W_{BF})$	Both > 30%	High
High-energy Coupled	> 4%	$< (2\,L_D + W_{BF})$ or $< (L_D + W_{BF})$	Both or at least one > 30%	High
High-energy Open	> 4%	$> (2\,L_D + W_{BF})$	Both or at least one > 30%	High
Moderate-energy Confined	0.1-4%	$< 7\,W_{BF}$	Variable	Moderate
Moderate-energy Unconfined	0.1-4%	$> 7\,W_{BF}$	Variable	Moderate
Canyon	Variable	$> 3\,W_{BF}$	> 70%	Moderate to High
Gorge	Variable	$< 3\,W_{BF}$	> 70%	Moderate to High
Glacial Trough**	< 4%	$> (2\,L_D + W_{BF})$	~ 10-% initially steepening to > 30%	Moderate to Low
Low-energy Floodplain	< 0.1%	$> 7\,W_{BF}$	Generally < 30%	Low

L_D – length of debris runout W_{BF} - width of channel at bankfull stage

** Defined as valleys with the given characteristics, lying above the elevation of the most recent glacial activity

PEER REVIEW AND EXPERT INPUT TO RIPARIAN FLOW-ECOLOGY RELATIONSHIPS

We held an Expert Panel Riparian Workshop on February 25, 2010 to peer-review completed work and to provide guidance on future efforts. One of the aims of which was to seek expert input on appropriate geomorphic classes for riparian cottonwood forest. Geomorphic classes with steep slopes and small stream size would not support significant stands of cottonwood. In addition to the reduced occurrence of cottonwood in canyons, flow is less important for recruitment here because rockslides are probably more important drivers of recruitment (canyons are highly coupled to side slopes). Those classes with slopes <4% and uncoupled with side slopes were therefore considered candidate classes. The magnitude of flow events required for successful recruitment is a product of geomorphic context because smaller floods are better able to rework the finer sediment of meandering reaches (Rood et al. 2007). Braided rivers are generally absent from the Colorado basin, and are not captured by the proposed riparian methods because different ecological processes occur (e.g. braided river systems can respond to flow regulation with increased cottonwood forests as the channel narrows; Scott et al. 1996, Marston et al. 2005, Graf 2006).

Within the suitable elevation and geomorphic contexts for cottonwood, there will be reaches where *P. deltoides* are absent. For example, floodwalls (even low ones) cut off the riparian zone from the river and can render otherwise suitable geomorphic classes unsuitable (see Table 9 in Hauer et al. 2002). We considered it unlikely that these alterations could be mapped reliably across the landscape, or that they were particularly prevalent across the Colorado watershed. Channels that have incised (e.g. sediment starvation from impoundment) also abandon the floodplain, so are not as suitable as indicated by broad geomorphic setting. Heavy browsing and felling of cottonwood can also eliminate cottonwood from otherwise suitable habitats (Auble and Scott 1998, Beschta 2003, Samuelson and Rood 2004). At this point, societal values could also be overlaid in terms of where conservation of cottonwood forest is a priority. The basin roundtable may choose to consider these additional non-geomorphic constraints for site specific evaluations or priority areas, but these are not dealt with here at a watershed scale.

RISK CLASSES

Flow-ecology relationships are used to assess potential changes in the status of flow-related attributes such as fish or riparian vegetation. In the Watershed Flow Evaluation Tool, we use "risk classes" as an indicator of the probability that the status of a given attribute will change relative to a reference status as a result of flow management. The hydrologic regime of a stream or river is a "master variable" governing the condition of species and ecosystems (Poff et al. 1997), yet other factors (land use, water quality, etc.) can also affect the status of river attributes. As such, risk classes are not deterministic, that is a "high" risk class does indicate that the attribute will for certain be in a state that is far-removed from the reference state, but it does imply that the chances of the attribute being farther removed are higher because of flow alteration.

Demarcation of risk classes is both a data-driven science process and a social process. The science process uses patterns in data, understanding of mechanisms of ecological function, and ecological principles to demarcate class. The social process adjusts the scientists assessment of risk classes to factor in values of those stakeholders who are applying the flow-ecology relationships with thresholds that better reflect acceptable levels of biotic alteration.

METHODS USED TO DEVELOP RIPARIAN FLOW-ECOLOGY RELATIONSHIPS

Wilding and Poff (2008) developed a flow-ecology relationship for riparian areas in Colorado below 9600' elevation. This relationship is still recommended for cottonwoods in confined geomorphic settings. In this report, two new flow-ecology relationships are developed specifically for cottonwood in unconfined valleys, including one for abundance of adult cottonwood and one for cottonwood recruitment.

Since Wilding and Poff (2008) was published, a dataset has become available that focused on cottonwood and used standardized survey methods applied across many sites (Merritt and Poff 2010). This dataset was employed here to derive the two new flow-ecology relationships. Merritt and Poff (2010) developed relationships for cottonwood and tamarisk, but the flow metric was deemed incompatible with the WFET (requires instantaneous flow data). As such, we re-analyzed the data using flow metrics that can be derived using a daily flow time series from StateMod (CDWR and CWCB 2009) followed by analysis of this time series with the Indicators of Hydrologic Alteration (IHA) software package (Version 7.1.0.10, Richter et al. 1996).

FLOW DATA

Merritt and Poff (2010) used a multivariate indicator of hydrologic alteration termed the IFM (index of flow modification). This index condensed various metrics for peak flow and low flow in terms of their deviation from unregulated conditions for each site[3]. The index performs well in representing flow alteration while dealing with collinearity (non-independence) among the various flow metrics, but is not directly interpretable in terms of flow units. It also uses component flow metrics that are not compatible with StateMod (e.g. instantaneous return period flows, cf. daily time series generated for StateMod nodes). So for the present study, cottonwood data from the Merritt and Poff (2010) dataset was re-analyzed using flow metrics that can be produced using StateMod, and that relate directly to the flow management questions being asked of this investigation.

As an initial step, data were obtained from David Merritt (USFS) providing the USGS gage numbers used, demarcation of flow data into pre- and post- alteration (normally temporal, but occasionally spatial), and a broader range of flow metric data. The record for unregulated rivers was divided in half for calculation of a “pre“ and “post” period comparable to regulated rivers, thereby allowing for natural variability in streamflow over long periods of time (i.e. non-stationary climate). Streamflow data for the relevant sites were then downloaded from the USGS website to enable a new analysis. One site was omitted at this point because daily data are no longer available (Rio Grande USGS 08332010), presumably because of poor quality (estimated alteration here was extreme at 170% increase for the 90 day maximum, in deviation from nearby gages).

Years with missing data (>10 consecutive days) were omitted from the analysis, which typically only affected the first and last year of record, with the revised record summarized in Table 2. The long periods of flow record used by Merritt and Poff (2010) meant that omitting data-short years had little effect on flow metrics for most sites (Figure 1). The largest deviations in metrics were for the Rio Grande (deviants from 1:1 line in Figure 1). The gages used and periods of pre and post alteration were revised for the Rio Grande following the recommendations of a separate hydrologic analysis that specifically examined hydrologic alteration for the Rio Grande (Wesche et al. 2005). Their recommended divisions of the flow record were therefore followed (1942-70 for pre-Cochiti Dam, and 1975-2003 for post). Changes were also made to the selection of pre and post records for Rio Grande sites RG1 and RG2 (vegetation study sites). The

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[3] Merritt and Poff (2010) performed a Principal Components Analysis on 8 flow metrics, from which the significant axes were used to calculate euclidean distance of each site from the centroid of unregulated rivers.

USGS gages 08361000 and 08362500 were used as the post alteration gages for sites RG2 and RG1 respectively (1975-2002 – post Conchiti period from Wesche report). Following the Wesche et al. (2005) recommendation, the USGS gage 08358500 for the period 1936-1958 was used as the pre-alteration record for sites RG1 and RG2 (cf. USGS gage 08358400 used as a spatial reference by Merritt and Poff 2010). This gage is at the same location as the gage used by Merritt and Poff (2010) (San Marcial) but has the advantage of predating Conchiti dam, as well as predating the flow division between a low flow conveyance and a flood channel at this site (now represented by USGS 08358300 & 08358400 respectively).

Omitting years with gaps in the flow record reduced the pre dataset for the Little Colorado at Woodruff (USGS 09394500) to just one year of data, and closer examination revealed unlikely spikes in the data (e.g. rising from 33 cfs to 10,000 cfs in one day). A similar 24-hour spike in flow is seen in other years on the exact same date (November 27) and also several times on December 4. Given the date repetition, these may have been an end of year release from Lyman Reservoir or, coincidentally, one of several known dam bursts that occurred at this site (though no record of their dates was found). These unseasonably high flows were therefore omitted as erroneous. To better represent the pre-alteration flows, the data that are available were pieced together. Flows were averaged for each day of the year across the period 1905-1920. Most days had 5 years data (ranging from 3 to 6 days) providing an improvement over the one year of complete record available. An additional year of data was produced by synthesizing a flow record from a nearby gage with overlapping record:

$\mathit{USGS09394500} = 0.315.\mathit{USGS0938600}^{1.2249}$ $R^2 = 0.70$ for 1906-1907

The output of these revisions was a single average year of data that provided more robust flow metrics.

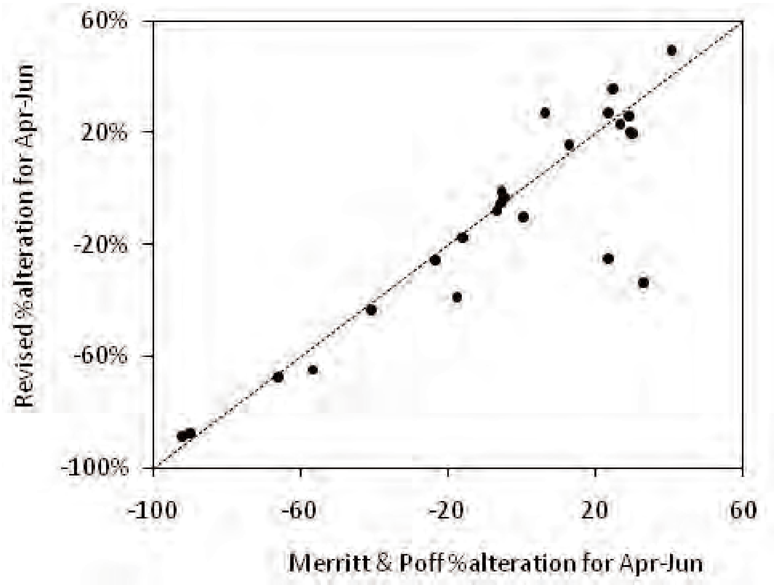

Figure 1. Comparison of revised and original flow statistics, comparing values used by Merritt and Poff (2010) and those recalculated for this investigation (omitting data short years and some change in gage sites used). The flow statistic being compared is the mean flow for April to June expressed as a percent alteration (post-pre/pre). The dashed line is a 1:1 line – the revised estimates that equal the original value will fall on this line.

Table 2. Hydrological record used to assess alteration of flow, including the USGS gage number, river and location, duration of pre- and post-alteration, intervening years that were omitted due to missing data ("Omit" column) and the vegetation monitoring sites that each gage record was applied to. See Merritt and Poff (2010) for additional information.

USGS Gage	River	Pre-alt.	Post-alt.	Omit	Vegtn. site no.
08330000	Rio Grande, Albuquerque, NM.	1943-1970	1975-2002		RGM7-1, RGN1-1, RGS1-5
08332010	Rio Grande, Bernardo Floodway, NM.	1958-1974	1975-2002		RG3 (omitted)
08361000	Rio Grande, Elephant Butte Dam, NM.	1936-1958 USGS 8358500	1975-2002		RG2
08362500	Rio Grande, Caballo Dam, NM.	1936-1958 USGS 8358500	1975-2002		RG1
08383500	Pecos River, Puerto De Luna, NM.	1939-1978	1979-2002		PEC-1 & 2
08384500	Pecos River, Sumner Dam, NM.	1913-1936	1937-2002	1926	PEC-3 to 5
09095500	Colorado River, Cameo, CO.	1934-1963	1964-2004		GJ-665 & 666
09128000	Gunnison River, Gunnison Tunnel, CO.	1911-1965	1966-2003		GUN-1 & 2
09163500	Colorado River, State Line, CO.	1952-1966	1967-2004		GJ-667 to 670
09169500	Dolores River, Bedrock, CO.	1918-1983	1984-2003	1971	DOL-2
09177000	San Miguel River, Uravan, CO.	1955-1978	1979-2003	1996	SM-1
09180000	Dolores River, Cisco, UT	1952-1983	1984-2003		DOL-1

09251000	Yampa River, Maybell, CO.	1917-1962	1963-2004		YAM-1 to 3
09384000	Little Colorado River, Lyman Lake, AZ.	1941-1970	1971-2003		LCR-34 to 35
09388000	Little Colorado River, Hunt, AZ.	1930-1949	1950-1972	1934, 1940	LCR-28, 29 & 32
09394500	Little Colorado River, Woodruff, AZ.	1905-1920	1930-2003	see report	LCR-15, 20 & 21
09402000	Little Colorado River, Cameron, AZ.	1948-1985	1986-2003		LCR6 & 10
09429100	Colorado River, Palo Verde Dam, AZ.	1957-1968	1989-2003		LC-T1 to T9, LC-T11 to T16
09431500	Gila River, Redrock, NM.	1931-1955	1963-2002		GILA1
09504000	Verde River, Clarkdale, AZ.	1916-1920	1966-2003	1917	VER-1 & 2
09506000	Verde River, Camp Verde, AZ	1935-1989	1990-2005		VER-3
09511300	Verde River, Scottsdale, AZ.	1962-1982	1983-2003		VER-6 & 7
10327500	Humboldt River, Comus, NV.	1895-1947	1948-2002	1910	HUM-1 to 5
10335000	Humboldt River, Rye Patch, NV.	1900-1932	1936-2002	1910, 11, 17 & 28	HUM-6 & 7
10351600	Truckee River, Derby Dam, NV.	1919-1957	1960-2002		TR-1 & 2

Following the Expert Panel Riparian Workshop several revisions to the draft riparian assessment were initiated. The first of these was a revision of flow metrics for predicting riparian response. Concerns were raised that relationships with annual floods may be a statistical artifact (see Baker 1990 for rationale). It was suggested that a flood peak with a return period of 3-5 years was more mechanistically linked to cottonwood recruitment and therefore population success, compared to annual floods (see Bradley and Smith 1986, Scott et al. 1996, Mahoney and Rood 1998, Rood et al. 2007). The Merritt and Poff (2010) analysis used instantaneous annual maxima series to generate 2, 10 and 25 year return period flood magnitudes. This cannot be generated by StateMod which is based on daily average data (not instantaneous flow). Following suggestions from the expert panel, additional metrics were calculated and analyses were done to compare various flow metrics based on a daily time-step to an instantaneous 5 year return period flood. The flow metrics used in this report are described in Table 3.

RE-ANALYSIS OF MERRIT AND POFF'S (2010) COTTONWOOD ABUNDANCE DATA

Abundance of cottonwood was assessed by Merritt and Poff (2010) as the proportion of plant occurrences in a series of transects. A 200 m long reach of river was selected and at every meter increment adult cottonwood occurrence (presence/absence) was observed for a perpendicular transect that ran across the entire floodplain. This provided 200x1 m wide transects from which to calculate %abundance, therefore:

% abundance = the proportion of 1m wide transects containing 1 or more adult cottonwood.

The reaches were replicated every 0.5 km. Analysis of the response of adult cottonwood abundance to flow alteration used quantile regression, following the methods stated in the original WFET report (Wilding and Poff 2008). These are restated here for completeness.

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-metric (e.g. peak-flow), but other variables (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.

Table 3. Flow metrics used in this report. Metrics calculated by Merritt and Poff (2010) are indicated by a asterisk. Note: instantaneous values are not StateMod compatible.

Flow metric	**Description**
Instantaneous 2, 5, 10 and 25 year return period*	Instantaneous annual-maximum peak flows for 2, 5, 10 & 25 years (flows with annual probability of exceedence of 0.50, 0.20, 0.10 & 0.04). The Pearson Type III frequency distribution was fit to the logarithms of instantaneous annual peak flows. Used PeakFQ software. Calculate flow for pre-alteration period, then repeat for post alteration. Percent flow alteration calculated in Microsoft Excel ([pre-post]/pre).
Daily 5 & 10 year return period	Daily series annual-maximum peak flows (Oct-Sept water year) for 5 and 10 year return events years (flows with annual probability of exceedence of 0.20 & 0.10). Calculated using IHA software by changing the EFC small flood return period from 2 to 5 years to generate a pre-alteration value (output under SCO worksheet as “EFC small flood minimum peak flow"). IHA appears to use a Weibull plotting position: P = rank/(n+1). The post-alteration value was then produced using a single period analysis constrained to post-alteration data. Percent flow alteration calculated in Microsoft Excel ([pre-post]/pre).
April-June average*	Mean flow for the April-June period is calculated for each year using IHA software, then averaged across years separately for both pre and post alteration periods. Percent flow alteration calculated in Microsoft Excel ([pre-post]/pre).
monthly average for April, May, June and July	As per April-June average, but calculated individually for each month.
1-day maximum	Annual maximum flow from the daily flow series (Oct-Sept water year) calculated using IHA software. This is then averaged across years separately for both pre- and post-alteration periods in Microsoft Excel. Percent flow alteration calculated in Microsoft Excel ([pre-post]/pre).
3-day, 7-day, 30-day and 90-day maximum	As per 1-day maximum, but annual maximum flow series is calculated as a moving average over 3, 7, 30 and 90 day periods instead of 1-day (i.e. the actual period of averaging is allowed to vary between years and sites).
Wet year 1-day, 3-day, 7-day, 30-day and 90-day maximum	In Microsoft Excel, wet years were identified as those exceeding the 70%ile MAF (threshold calculated separately for pre-and post-alteration). The annual maxima series (1, 3, 7, 30 and 90 day moving average) is then reduced to wet years only, and flows averaged across wet years separately for both pre- and post-alteration periods. Percent flow alteration calculated in Microsoft Excel ([pre-post]/pre).

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 (LAD - 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. The 90% quantiles were judged as representing the upper-bound response adequately. The necessity of transformations was investigated, 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 (positive or negative), not their magnitude. The permutation version uses an F statistic with its sampling distribution approximated by permutation (Cade et al. 2006), with 5000 permutations used here.

Re-analysis of Merritt and Poff's (2010) cottonwood recruitment data

Recruitment of cottonwood was investigated using the binary recruitment data from the Merritt and Poff (2010) dataset. The presence of 2-5 year old saplings was recorded when surveying each 200 m long reach, producing a presence/absence record for each reach (cf. %abundance per reach for adult cottonwood). The quantile regression analysis used for adult cottonwood is therefore not applicable to the recruitment data. The analysis by Merritt and Poff (2010) employed the IFM (index of flow modification) to predict recruitment response, based on a mixed effect logistic regression model.

The purpose of the analysis was to select alternative flow metrics to the IFM that are compatible with StateMod (i.e. derived using daily time series data). A subset of informative flow metrics was selected based on results from the adult cottonwood analysis and riparian workshop (instantaneous 10-year return period flow, daily series 5-year return period flow, maximum 90-day flow, and wet year maxima – 1, 7 and 90 day, described in Table 3). A logistic Generalized Linear Model analysis was then run to further narrow the list of candidate flow metrics. Using AIC (Akaike's Information Criterion), the 5-year return period flow and wet year 90 day maxima were selected as the most informative flow alteration metrics (Table 4).

Table 4. A logistic Generalized Linear Model was run for each of the following flow alteration metrics as predictors of cottonwood recruitment. The wet year 90 day maximum and 5-year return period flow (daily series) were the best predictors based on AIC (smaller better). The multi-metric based IFM (index of flow modification) from Merritt and Poff (2010) is also tabulated for comparison (modeled for this table using the same sites as the other metrics). A lower AIC value and a lower p-value both indicate a better model.

	AIC (smaller better)	P-value of coefficient Pr(>|z|)
90 day max	113.0	0.0589
10 yr return flow (instantaneous)	111.7	0.0284
wet year 7 day max	109.6	0.0096
wet year 1 day max	106.1	0.0023
5 yr return flow (daily)	**105.7**	0.0018
wet year 90 day max	**103.6**	0.0010
IFM	*93.6*	*0.0002*

Maxent was used to model the response of cottonwood recruitment to flow alteration (Dudik et al. 2010). Maxent attempts to estimate the most uniform or spread-out probability function (i.e. the distribution with maximum entropy), subject to constraints that are determined by the environmental data. In effect it makes no assumptions about the distribution of, in this case, recruitment beyond the flow constraints we can observe. It is a non-linear method that follows Bayesian principles in deriving an appropriate probability distribution function from the dataset (Phillips et al. 2006), rather than assuming that commonly used probability functions will be adequate. The model settings used included a regularization multiplier of 1, bootstrap evaluation with replacement for at least 50 model replications and with presence sites added to background (otherwise using defaults). Because absence sites were used as background data (termed “target-group” background), the model is expected to achieve better predictions than a presence-only analysis would with random background reaches, as demonstrated by Phillips and Dudík (2008). The AUC statistic was used to evaluate Maxent model performance. This measures the area under the receiver operator curve, with a value of 1 ideal and values <0.5 indicating predictions no better than chance.

The relatively small number of occurrences (22 reaches with recruitment observed) increases the importance of the method used in determining predicted response to flow alteration. Maxent was used to re-assess the data because of its strength in dealing with small numbers of occurrences and lack of assumption about the shape of the response (Pearson et al. 2007, Phillips et al. 2006, Phillips and Dudík. 2008). This method does not account for the nested sampling design used by Merritt and Poff (2010) (cf. NLME models), instead considering each reach individually. So the two methods were compared (NLME logistic regression & Maxent) using recruitment response to IFM (index of flow modification). There were some differences between NLME logistic regression and Maxent predictions (Figure 2). On average, Maxent predicted slightly higher occurrence at intermediate flow alteration (IFM 0.2-0.5) which is also the range with greatest variability (the predictions of each replicate model depends on which sites are included). The lower bound (-1 standard deviation) of the Maxent response is closest to NLME predictions overall (Figure 2). Certainly Maxent appears a valid method for investigating the response of recruitment to alternative flow metrics, especially given the flexibility of the response function.

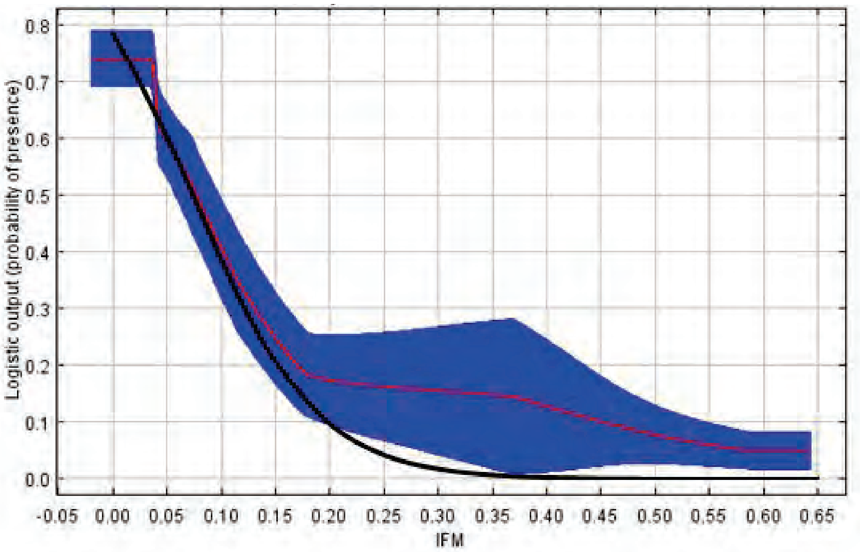

Figure 2. Probability of cottonwood recruitment in response to IFM (index of flow modification), comparing the predictions from NLME logistic regression (black line) to Maxent predictions (red line, with blue area ± 1 standard deviation generated from 50 bootstrap iterations). AUC = 0.806.

RIPARIAN RESULTS

RE-ANALYSIS OF THE MERRITT AND POFF (2010) DATA

Flow-ecology relationships for unconfined geomorphic settings: adult cottonwood

Among the metrics used by Merritt and Poff (2010) to describe peak flow, the 25 year return-period flow (instantaneous) had the highest R^2 value, which means it explained more of the variation in the data. However, as noted above, instantaneous values cannot be derived from StateMod. Therefore, additional flow metrics were calculated for this investigation to provide measures of peak flow that could be derived using StateMod (described in Table 3).

Cottonwood forest does not require high flows every year in order to achieve adequate recruitment. Therefore the flow data were re-analyzed using only wet-years. A wet-year was delineated as exceeding the 70^{th} percentile mean annual flow. The pre-alteration percentile cannot be applied post alteration because regulation can reduce the chance of the threshold being exceeded (i.e. the number of wet years will be underestimated). In the absence of a reliable indicator of natural wet years, we used the post alteration 70th percentile, which is still indicative of precipitation assuming that flows are somewhat uniformly altered between years (or at least between wet-years). Each flow metric was then averaged only across wet years and compared pre- and post-alteration. An additional two sites were omitted from this analysis due to insufficient replication of wet years (USGS 09394500 & 09504000). Note that the quantile regression analysis was constrained to sites with reduced flows (i.e. only sites with flow alteration ≤ 0) as we are primarily concerned with flow *reduction*.

Compared to pre-workshop analyses based on annual maxima, the wet year analysis (Figure 3) gave a less significant correlation for 30 day maximum (p-value increased from 0.095 to 0.18), but improved the significance for the 90 day maximum (p-value reduced from 0.027 to 0.015). All else being equal, we might have expected reduced significance of results from the wet year analysis because of the reduced dataset (70% less flow records), so the improvement exceeds expectations. All metrics approach a 1:1 relationship (1% flow reduction associated with 1% less cottonwood), especially if attributing more weight to the statistically significant relationships ($p<0.05$). The original WFET riparian analysis (Wilding & Poff 2008) also approached a 1:1 relationship, lending weight to this level of riparian impact from flow alteration.

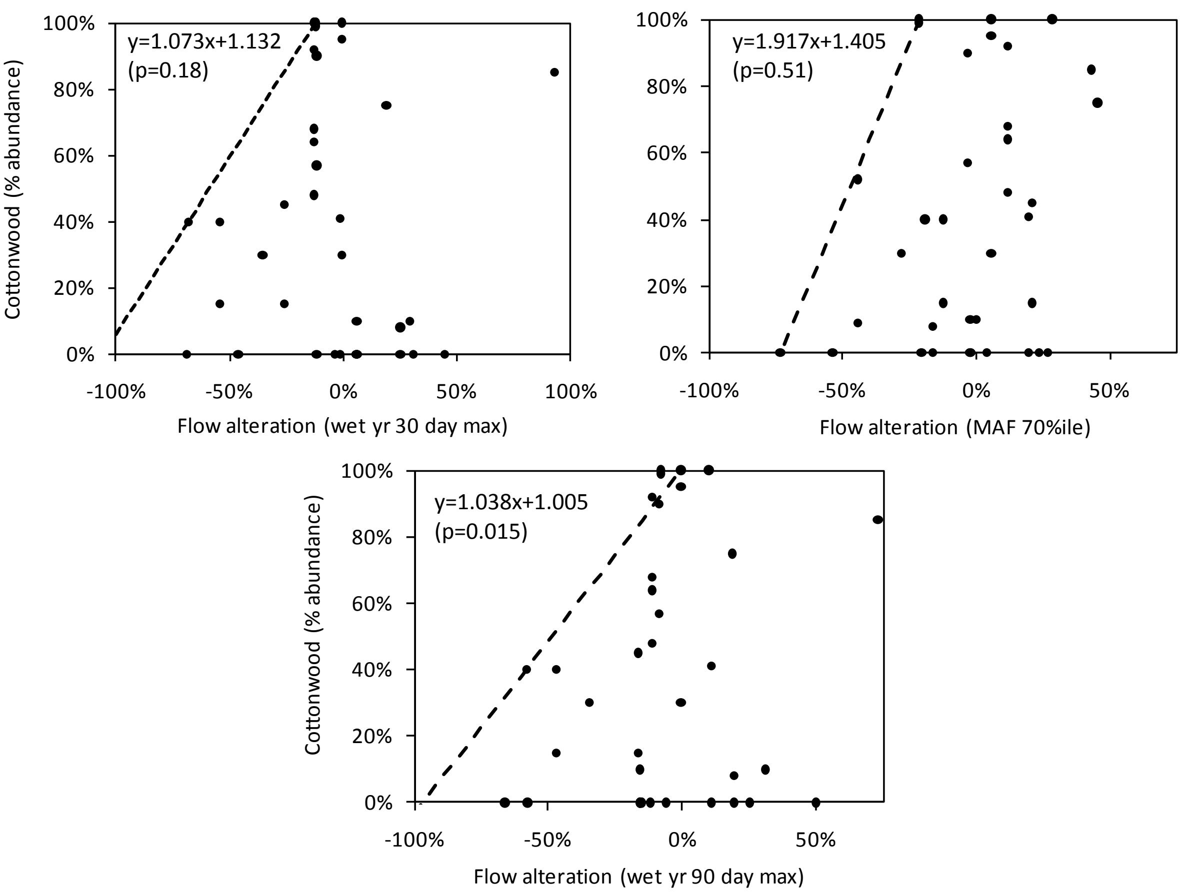

Figure 3. Cottonwood abundance response to peak flow alteration (30 day and 90 day max.) during wet years only (i.e. averaged over years exceeding 70%ile MAF). Necessary flow data was not available for three sites (USGS 08332010, 09394500, 09504000), hence were omitted. Alteration of the 70 percentile MAF (mean annual flow) is also presented (one datapoint at 143% alteration and 0 cottonwood is not shown on the MAF plot to achieve consistent axes). Note that all of these charts are comparable, indicating moderate to strong correlation among these flow metrics, but the 90 day maximum provides the best model.

The best predictor of a 5-year return period flood magnitude was investigated following recommendations from the Riparian Workshop. The instantaneous 5-year return period flood magnitude calculated by Merritt and Poff (2010) was used as the target metric. The daily series 5-year return period flow magnitude (produced using IHA software) gave the best correlation with the instantaneous estimate for the 5 year return period flow (also for the 10 and 25 year instantaneous flow). The next best correlate was the wet year 90-day maximum (Figure 4).

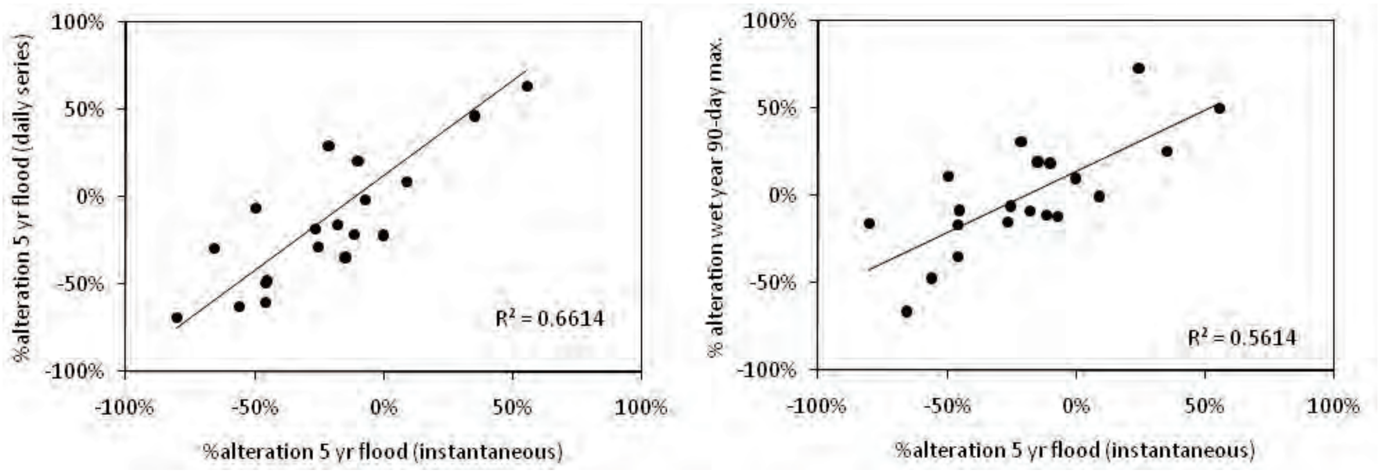

Figure 4. Correlation of two IHA metrics (daily series 5-yr return period flood and wet year 90-day max.) with the instantaneous 5 year return period flood.

Unfortunately the daily series 5-year return period flow is a poor predictor of cottonwood abundance, along with the wet-year 1-day and 7-day maxima (Figure 5). Visually, an underlying response can be seen (Figure 5), but the outliers are too pronounced to allow calculation of a valid relationship (p = 0.5). The instantaneous 5-year return period flow provided a relationship more consistent with other metrics, but was not significant (p=0.12).

The recommended function for evaluating the risk of flow alteration effects on cottonwood abundance is therefore based on the wet-year 90-day maximum flow. A response function was derived as the 90% quantile of the Merritt and Poff (2010) abundance data for adult cottonwood. The final recommended equation and risk classes are presented in the Executive Summary and need not be repeated here.

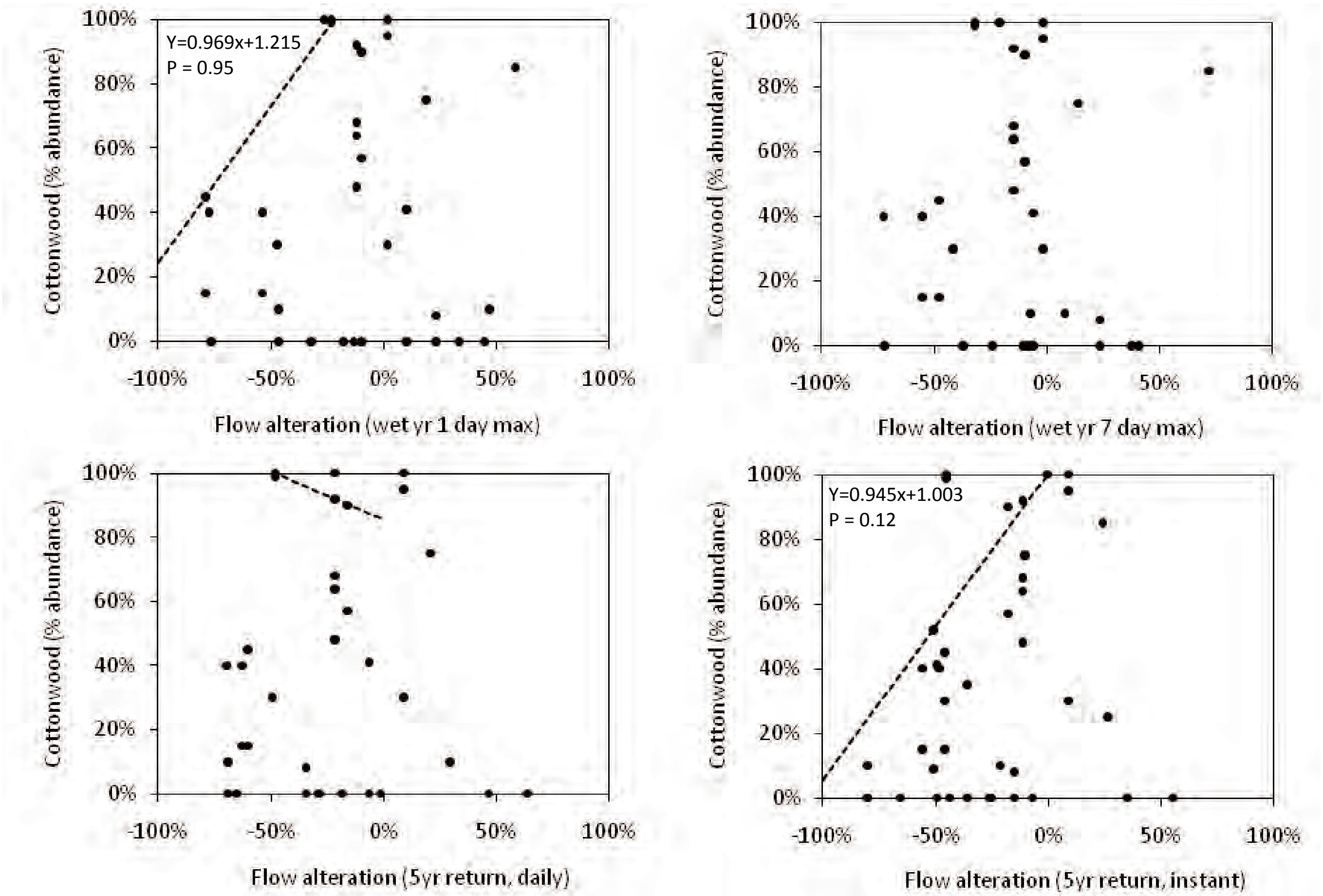

Figure 5. Adult cottonwood response to peak flow alteration. This plot includes two "wet year" average maxima (1-day and moving 7-day average), the daily series 5-year return period maxima and instantaneous 5-year return maxima. The dashed lines are 90% quantiles fit to the data (y=100% for 7-day).

Flow-ecology relationships for unconfined geomorphic settings: cottonwood recruitment

Recruitment of cottonwood was investigated using the binary recruitment data from the Merritt and Poff (2010) dataset. The occurrence of 2-5 year old saplings was recorded for each reach, hence this is a presence/absence dataset (cf. %abundance data for adult cottonwood). Two flow metrics were selected, as alternatives to the IFM, based on Statemod compatibility and predictive strength (wet year 90 day max, daily series 5 year return flow – see methods).

Maxent was used to analyze the data because of its strength in dealing with small numbers of occurrences and lack of assumption about the shape of the response (Phillips et al. 2006, Phillips and Dudík 2008). Using the wet year 90 day maximum predicts a reduced probability of recruitment when flow is reduced from natural, but not at sites with augmented flows (Figure 6). The other StateMod compatible flow metric (5 year return flow) was similar in the general form of the response to the 90-day max, with declining recruitment at reduced flows and stable recruitment under augmented flows (Figure 7). But the predictive performance using 5 year return flow is not as good and the predictions more variable (AUC=0.72, cf. 0.7 lower cutoff used by Phillips and Dudík 2008).

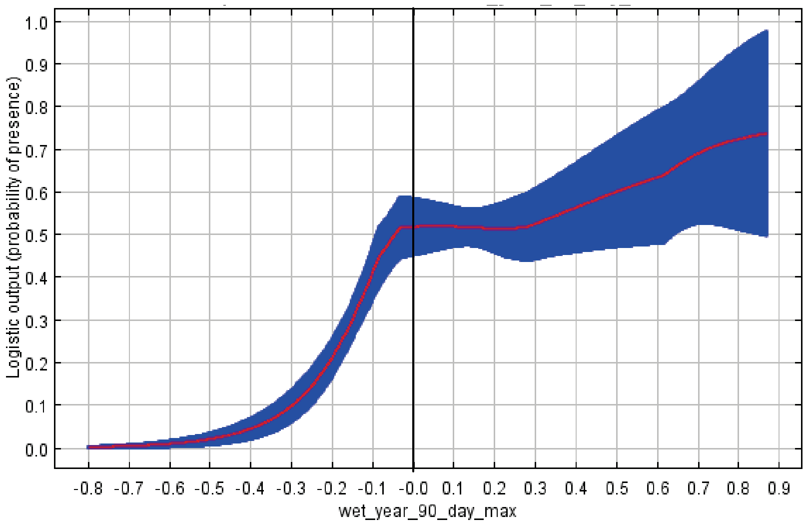

Figure 6. Probability of cottonwood recruitment in response to alteration of wet year 90-day maximum flow predicted using Maxent (mean response is the red line, with blue area ± 1 standard deviation generated from 100 bootstrap iterations). An unaltered flow is 0 on the x-axis (-0.5 represents a 50% reduction in flow). AUC = 0.775.

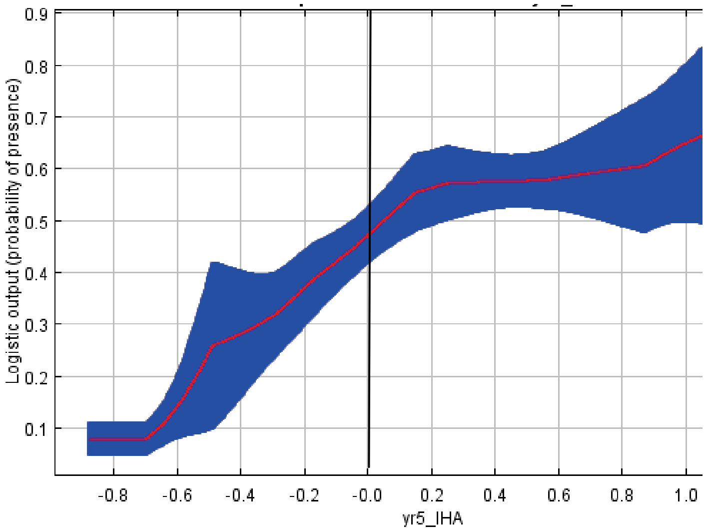

Figure 7. Probability of cottonwood recruitment in response to alteration of 5 year return flow predicted using Maxent (mean response is the red line, with blue area ± 1 standard deviation generated from 50 bootstrap iterations). An unaltered flow is 0 on the x-axis (-0.5 represents a 50% reduction in flow). AUC = 0.719.

The recommended function for evaluating the risk of flow alteration effects on recruitment of cottonwood is therefore based on the wet-year 90-day maximum flow. A polynomial response function was derived from the Maxent output to simplify implementation using post-processing in Microsoft Excel. This polynomial function adequately reproduces the flow-reduction portion of the model and is more easily applied than the multiple functions (or features) generated by Maxent. The final recommended equation and risk classes are presented in the Executive Summary and need not be repeated here.

Comparison of adult cottonwood and cottonwood recruitment curves

A comparison of the cottonwood curves for binomial recruitment versus adult abundance is therefore worthwhile in considering their application. Certainly predicted recruitment declines more steeply than the predicted abundance of adult cottonwood in response to reduced wet year 90 day maximum flows (Figure 8). Recruitment is needed to sustain cottonwood forest, but adult cottonwoods are present at sites that have a low chance of recruitment. Arguably, this could be interpreted as meaning some level of abundance of adult cottonwood can be supported by low rates of recruitment (e.g. 50% adult abundance was sustained by 4% of natural recruitment where flows are reduced by 50%). Alternatively, sites experiencing significant flow alteration may not be experiencing adequate recruitment and those adults that were observed are simply the remaining fraction of a forest that is slowly dying out. Certainly the recruitment function provides a more protective evaluation of risk of effects from flow alteration, with more certainty that the function describes flows that sustain cottonwood forest in the long term.

We anticipate that the wet year 90 day maximum flow is mechanistically linked to critical recruitment processes for cottonwood. The wet year 90 day maximum does not measure the duration of the effective discharge (flows that are effective in mobilizing sediment to create bare colonization sites, sensu Richter and Richter 2000), but this flow metric is expected to be correlated with the effectiveness of flood events. Nor does it capture the timing of flows relative to cottonwood seedfall. Equally so, representing 10 years of data with one 15-minute interval (instantaneous 10 year return flow) or 5 years of data with 1 day of recorded flow (daily series 5 year return flow) falls short of capturing all components of the flow regime necessary for recruitment. Results here suggest the wet year 90 day maximum does the best job, out of the individual metrics considered, of indicating the suitability of the broader flow regime. It is therefore an indicator of flow adequacy rather than a description of the complete flow requirements of riparian cottonwood. The latter would be required for site-specific flow prescriptions (see Mahoney and Rood 1998).

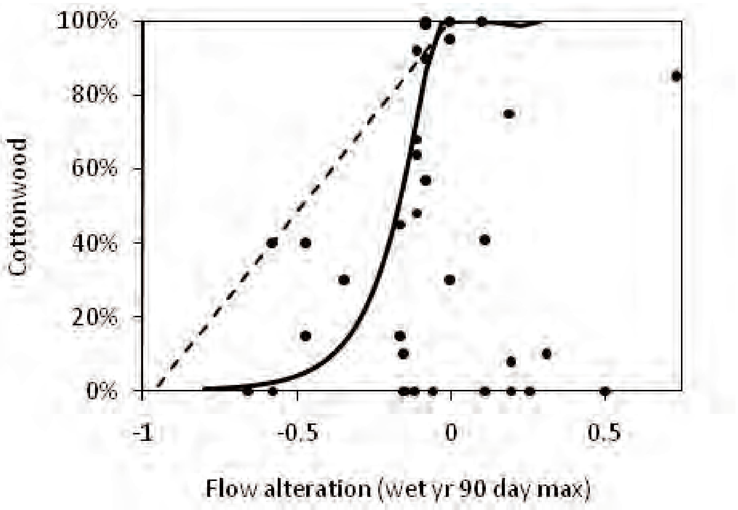

Figure 8. Comparison of adult cottonwood (dashed line and black dots, %abundance) to cottonwood recruitment (solid line, % of natural recruitment probability) in terms of their response to alteration of the wet year 90 day maximum flow.

COTTONWOOD FLOW-ECOLOGY RELATIONSHIP FOR CONFINED SETTINGS

The Merritt and Poff (2010) derived flow-ecology relationships are valid only in unconfined valleys, so a separate relationship is recommended for steeper, more confined geomorphic settings. Peer-reviewed research indicates that recruitment and growth of cottonwood in confined settings is related to flow, but the mechanisms of this relationship differ from unconfined settings (e.g. Roberts 1999, Stromberg and Patten 1991). There is some consensus in the literature that a less frequent flood drives recruitment in confined settings, typically in the order of 10-15 years recurrence, regardless of species (Table 5). Seedling establishment occurs more often in confined rivers (3-5 years), as it does in unconfined rivers. But survival to reproducing adults (i.e. recruitment) is unlikely from these smaller events in confined settings, so the bigger floods (10-15 yr return) are more of a necessity. Confined valleys are generally more prevalent at higher elevations where the climate is cooler and wetter. This reduces the dependence on receding flows to provide moisture for seedling growth (Polzin and Rood 2006), and large trees may instead source water from deeper groundwater originating from hillslopes (Dawson and Ehleringer 1991).

Confined valleys at lower elevations will be drier, and hence flow recession rates will be more critical for cottonwood here. This is a relatively harsh environment for cottonwood establishment, and it is therefore expected to support sparse cottonwood stands. The faster growth rate of *P. deltoides* seedlings may increase their chance of success at lower elevations, compared to narrowleaf cottonwood seedlings (Kalischuk et al. 2001). Seedlings are expected to be very dependent on surface water in this setting (Dawson and Ehleringer 1991), compared to root suckers from narrowleaf cottonwood that benefit from deeper groundwater (Krasny et al. 1988).

Table 5. Cottonwood stand recruitment data from confined rivers. Data were sourced from each article where available, otherwise were estimated from aerial photos in Google Earth.

Study	species	Valley slope	Confinement	Flood recurrence interval for recruitment	Flow alteration
(Scott et al. 1997, Auble and Scott 1998)	*P. deltoides*	0.05%	confined (valley width ~3x bankfull width)	9.3 years for adult recruitment from seed.	"attenuated peak flows by 14-23%"
(Baker 1990)	*P. angustifolia*	2%	Canyon (valley width ~2x bankfull width, colluvial deposits in channel)	10-15 years for adult recruitment, 3.4 years for seedlings that presumably failed.	"unregulated"
(Polzin and Rood 2006)	*P. trichocarpa*	0.6%	Confined (valley width 2 to 6x bankfull width)	100 yr for seedling recruitment; weak flood association for root suckers.	"run of river dam"
(Samuelson and Rood 2004) montane results	*P. trichocarpa*	3%	Confined (sinuosity <1.5)	5 yr for root sucker recruits, >50yr for seedling recruits.	Unregulated

In the absence of new data to describe cottonwood response in confined settings, *the recommended function for evaluating the risk of flow alteration effects on cottonwood in confined settings* is Method 7 from Wilding & Poff (2008). The final recommended equation and risk classes are presented in the Executive Summary and need not be repeated here.

Additionally, we recommend evaluating alteration of the 1-in-10 year 90-day maximum flow (i.e. direct consideration of degree of flow alteration). Large floods are important for cottonwood recruitment in this setting. We cannot quantify the degree of risk associated with alteration of this flow metric, but it could at least be used to narrow down the list of sites where further investigation of effects may be justified (e.g. sites where the 1-in-10 year 90-day maximum flow is reduced by more than 10%).

FLOW-RESPONSE FOR WILLOW (SALIX SPP.)

Willows (*Salix* spp.) are a diverse genus, and belong to same family as cottonwood (Salicaceae). Most members of this family are riparian/wetland specialists (Karrenberg et al. 2002), and willow are no exception. Among Colorado's 30+ species of willow, nearly all grow in moist habitats of wetlands and/or riparian areas (Weber and Wittmann 2001a, b). In Colorado, willow ecosystems (termed willow carrs) are often dominant in broad valleys (including unconfined and glaciated valleys) with low valley slopes (<3%) in montane and subalpine settings (Patten 1998, Rocchio 2006). Flow-ecology relationships were investigated in this geomorphic setting for a subset of species (*S. planifolia, S. geyeriana, S. monticola*, and *S. petiolaris*) because they dominate montane and subalpine willow carrs in Colorado (Carsey et al. 2003) and are known to depend on floods (Woods and Cooper 2005, Cooper et al. 2006). Willow carrs were divided by Carsey et al. (2003) into two types: tall shrublands (e.g. *S. geyeriana, S. monticola* from 7,700-10,300 ft) and short shrublands (e.g. *S. planifolia* from 8,300 to 12,000 ft).

Establishment and growth of most willow species depends on interactions between hydrology, geomorphology and animals, with bare, moist surfaces formed by floods being particularly important to establishment of plants and high water tables being important for long-term survival and growth (Krasny et al. 1988, Naiman and Décamps 1997, Karrenberg et al. 2002, Woods and Cooper 2005, Cooper et al. 2006, Westbrook et al. 2006). These aspects of the ecology of willows, including their reproductive mechanisms and strategies are similar to other members of the plant family Salicaceae, including cottonwood (genus *Populus*). Among the Salicaceae, many species reproduce sexually (i.e. by seed) or asexually (e.g. sprouting from broken branches). The importance of one means of reproduction versus another varies based on species and physical setting (Krasny et al. 1988). Sexual reproduction by seedfall was observed to be dominant for riparian willow at higher elevation (Cooper et al. 2006). The species considered here are capable of asexual reproduction, but this is rarely observed as an origin of mature riparian stands. Asexual reproduction is more important in wetlands than riparian shrublands (including species that inhabit both environments), though we do not understand the mechanisms of this transition.

Recruitment is expected to respond more immediately to flow alteration, compared to aerial extent of willow shrublands, because willow are relatively long-lived (>40 years, Cooper et al. 2006, Wolf et al. 2007). Cooper et al. (2006) demonstrated that willow recruitment depends on flooding events to create appropriate surfaces and hydrologic conditions - processes that are in many ways similar to those mechanisms supporting cottonwood recruitment at lower elevations. In particular, smaller flood events (annual return) were associated with recruitment in meandering rivers (point bars left behind by meandering), larger floods for recruitment of abandoned channels (2-5 yr return) and infrequent floods for recruitment of abandoned beaver ponds (>5 yr return). Flow alteration can impact channel processes of wide valleys at high elevations, as demonstrated by Ryan (1997) in the headwaters of the Colorado River, and it follows that flow alteration could affect willow establishment, growth, and survival. A decline in willow extent following flow regulation was observed in Arizona and Montana (Lite and Stromberg 2005, Marston et al. 2005).

Floods and streamflow are important drivers of willow ecosystems and the general processes are assumed to be similar to cottonwood. But there are several other major drivers that can overwhelm the response of willow to flow alteration. Beaver are major drivers of willow shrublands, as well as riparian-stream ecosystems as a whole (Naiman et al. 1986, Cooper et al. 2006, Rocchio 2006, Westbrook et al. 2006, Wolf et al. 2007, Westbrook et al. 2010), acting as a major disturbance of riparian areas through flooding, vegetation clearing and as a modifier of channel response to floods. Beaver ponds are important for raising groundwater levels above and below the dam. The bare surfaces exposed by failed beaver dams are important recruitment sites for willow that often extend the zone of flood influence and, consequently, willow shrubland (cf. no beaver dam). Beaver activities cause channel avulsion, which also produces bare surfaces. Beaver affect sediment deposition, increasing the quantity and proportion of fines in soils to the benefit of willow (by producing soils with better moisture retention). The loss of floods can therefore be mitigated by beaver to some extent, as they provide an alternate source of disturbance and reduce the dependence of groundwater levels on stream flow. But this limits the disturbance to one source and creates a system that is very susceptible to other stressors, such as overgrazing. People may actively remove beaver for the purposes of development (e.g. agriculture, diversion schemes). The loss of beaver can also result in channel incision as the stream adjusts to a new regime of sediment and water retention (Wolf et al. 2007). Channel incision can result in floodplain abandonment by the stream and subsequent loss of willow recruitment.

Willow shrublands are associated with shallow groundwater (Krasny et al. 1988, Gage and Cooper 2004). In some settings, groundwater is recharged primarily from adjacent hillslopes, rather than the stream. High recharge rates can originate from deep glacial till, hillslopes with highly fractured rock and longer hillslopes, particularly those with low slopes that drain more slowly. Typically, the higher the elevation the higher the magnitude of hillslope discharge as a consequence of precipitation-evaporation patterns in Colorado (Patten 1998). Groundwater does not directly influence recruitment processes (such as meandering and point bar migration), but groundwater does affect biomass of existing vegetation (Dwire et al. 2009) and vulnerability to grazing effects (Peinetti et al. 2001). Also, substantial groundwater inputs can mitigate effects of diversions depletions by rapidly recharging the stream below a diversion, and these inputs can provide opportunity for beaver activities that lead to recruitment. The less water originating from hillslopes the more dependent willow will be on streamflow. Intermittently flowing streams reflect low groundwater levels, and therefore may not support willow. Beaver dams can raise groundwater levels (Westbrook et al. 2006), increasing willow success in intermittent streams and drier valleys.

Flow-ecology curves

As this review demonstrates, much research has established the basic mechanisms by which willows depend on the flow regime, geomorphic setting and beaver activity. Nonetheless, specific quantitative descriptions of flow dependence of willow recruitment has not received the same level of research effort as cottonwood (see Lite and Stromberg 2005, Marston et al. 2005). The flow-ecology relationship for cottonwood in unconfined geomorphic settings provides a good starting point because the same channel processes are involved. In particular we see seedling establishment associated with point bar migration and channel cutoffs regardless of whether cottonwood or willow are the dominant riparian species. The results from Cooper et al. (2006) indicate similar recruitment processes in this setting, with the "effective" flood for recruitment being 2-5 years. In addition to similar channel forming processes, the strategies for reproduction and growth are similar across many of the Salicaceae (Karrenberg et al. 2002). The similarity extends to the timing of seed rain for willow and cottonwood (Niiyama 1990, Mahoney and Rood 1998, Cooper et al. 1999, Gage and Cooper 2005), which reaches a maximum on the receding limb of snowmelt peak flow (Figure 9).

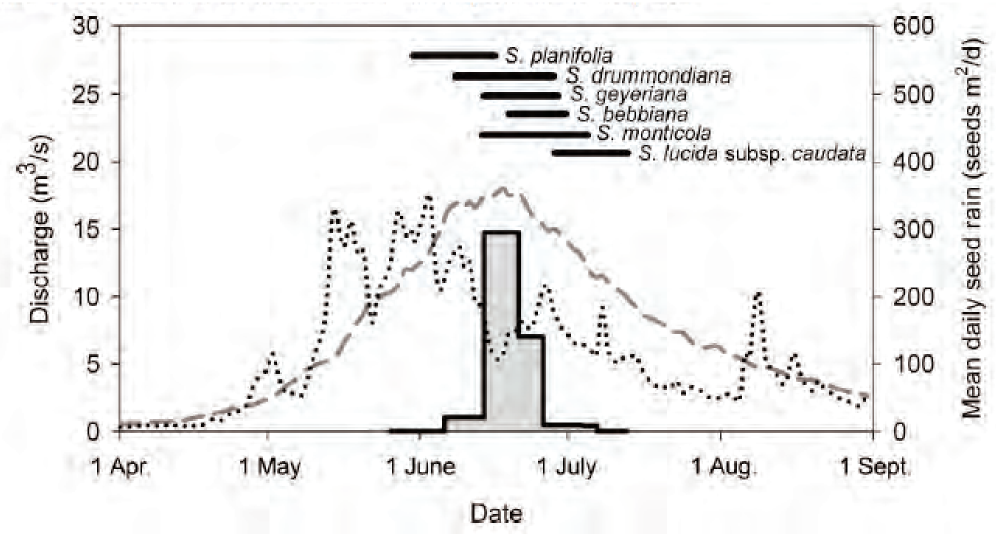

Figure 9. This figure, reproduced from Gage and Cooper (2005), describes the timing of willow seed rain (shaded columns) relative to snowmelt flow (dotted line – same year flow, dashed line – average flow). The seed-release period for individual willow species are also described by horizontal bars.

We can at least formulate hypotheses of the relationship between willow and flow alteration, and these hypotheses were developed into plots (Figure 10). These concern montane and subalpine willow shrublands in wide valley settings, including *S. planifolia, S. geyeriana, S. monticola*, and *S. petiolaris*. The first scenario (Scenario A) represents a largely intact system, with beaver widespread and low levels of grazing, clearance and developmental pressures. In this situation, beaver have the potential to mitigate much of the impact of flow alteration on disturbance regimes. The range of response for Scenario A is expected to vary depending on the degree of alluvial groundwater recharge from adjacent hillslopes (cf. streamflow recharge). High recharge from hillslopes is expected to offer some mitigation for the effects of flow alteration, because willow productivity/survival is less dependent on stream flow for groundwater recharge. As discussed previously, an absence of floods for Scenario A streams may support expansive willow cars, but is very susceptible to additional stressors.

Scenario B lacks severe grazing and developmental pressures (as per Scen. A), but also lacks beavers. In this scenario we expect willow shrublands to be most susceptible to flow alteration. Note that we do not expect the natural flow regime will be sufficient, in the absence of beaver, to maintain maximum potential for willow shrubland (i.e. willow maintenance is <1 at flow alteration of 0).

For Scenario C, direct pressure on willow from grazing and other development is high and beaver are expected to be largely absent as a direct or indirect consequence of development/grazing. In this scenario we do not expect to see extensive willow shrublands regardless of flow alteration (or lack of). Willow may be reduced to a narrow strip along the stream banks. Heavy grazing can trigger collapse of beaver-willow communities (Baker et al. 2005), with low groundwater levels increasing susceptibility to grazing effects (Peinetti et al. 2001).

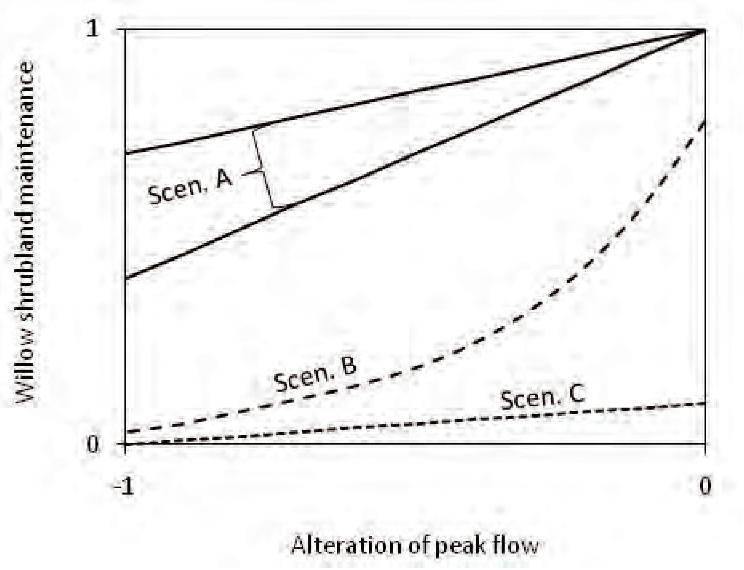

Figure 10. Hypothesized response of riparian willow to flow alteration under 3 scenarios. These concern willow shrublands in wide valley settings, including *S. planifolia, S. geyeriana, S. monticola*, and *S. petiolaris*.

Scenario A – Beaver present, with an upper and lower range of response depending on degree of recharge of alluvial groundwater from adjacent hillslopes (low hillslope recharge for lower line).

Scenario B – Beaver absent. High dependence on snowmelt floods for willow recruitment and ultimately for shrubland maintenance.

Scenario C – Heavy grazing and or clearing of willow.

These different response scenarios suggest that application of flow-ecology curves should be targeted at a subset of the wide, low-to-moderate gradient valleys >8000 feet. Where beaver are active (Scenario A), particularly where there are significant groundwater inputs, willow shrublands are less likely to show a dramatic decline in response to flow alteration. Therefore, consideration of willow response to flow alteration is a low priority in these locations. Flow-ecology relationships could be applied to both Scenario B and Scenario C. In Scenario B (limited grazing and development, but without beaver), willows are expected to be most sensitive to flow alteration. In Scenario C (human activities trump ecological processes), unaltered flow indicates the potential for healthy willow ecosystems, but the realized extent of willow shrublands is limited by other factors. Identifying streams that lack beaver (Scenario B and C) across the Colorado basin (wide valley, montane-subalpine) would allow targeted application of flow-ecology relationships where flow alteration is most likely to constrain the potential extent of willow shrublands.

Because the data available to describe flow-ecology relationships for willow are limited, *we do not recommend a quantitative function for evaluating the risk of flow alteration effects on* willow. However, alteration in peak flows can provide a basis for general inferences about risk to willows, using the conceptual relationships described above.

The flow metric that could be used to describe peak flow alteration also deserves consideration. The flow metric used for cottonwood (90-day maximum) may be too long because streamflow patterns are expected to be less important for post-germination survival of willow at high-elevations compared to cottonwood in semi-arid areas (Patten 1998). Temperatures are cooler and available moisture is expected to be higher above 8000 feet (both atmospheric humidity and soil moisture), so willows may tolerate being disconnected from the water table. Woods and Cooper (2005) observed a correlation between willow seedling survival and soil moisture within 3 weeks of the snowmelt peak (the "steep recession limb of the snowmelt hydrograph"), but not later in the year and little apparent benefit from supplemental irrigation. Additionally, the growing season is short at high elevations, which constrains the maximum duration per year of streamflow influence on plant growth. Therefore, the 30-day maximum flow may be a better indicator metric, compared to the 90-day maximum used for cottonwood. The 1-day maximum flow is likely correlated with the 30-day maximum flow and thus could be informative as an indicator metric. The return period of flow events that are associated with recruitment of willow (3-5 years, Cooper et al. 2006) are equivalent to that described for cottonwood in wide valley settings. The consideration of only wet-years (years exceeding the 70%ile mean annual flow) for cottonwoods could also be used for willow.

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Appendix G

Recreational Survey Report

AMERICAN WHITEWATER

Assessing Streamflow Needs for Whitewater Recreation in the Yampa/White River basin.

Integrating Overall Flow-Comparisons and Single-Flow Judgments to Define Low, Acceptable, and Optimal Flows for Whitewater Recreation

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Abstract:

Streamflow, or the amount of water in a river, affects the quality, quantity, and timing of river-related recreation, such as whitewater boating. This report describes flows that provide whitewater boating opportunities for various craft-types on targeted river segments in the Yampa and White River Basins in Northwest Colorado. American Whitewater conducted the study in 2011, at the request of the Yampa-White Basin Roundtable created under Colorado's Water for the 21st Century Act. In this study we used two approaches to assess the relationship between streamflows and recreation quality. An online survey collected information from 292 respondents who evaluated flows for whitewater boating on 17 Recreation Attributes[1] in the Basin. Respondent data was collected and organized to identify minimum, acceptable and optimum flows for whitewater boating, summarized by Flow-Evaluation curves describing the quality of boating opportunities for each measured stream-flow. Respondents also reported flows that provide certain recreation experiences or "niches", from technical low water to challenging high water trips. This report integrates the results of overall flow-comparisons with single flow assessments of recreation quality, to describe flows that provide whitewater recreation opportunities in the Yampa and White River Basins. This report provides a baseline set of information for whitewater recreation in the Yampa-White Basin that can be helpful when evaluating future water management actions, climate change analysis, or risk management strategies that impact streamflows.

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[1] National Inventory of Whitewater Rivers; American Whitewater. http://www.americanwhitewater.org/content/River/view/

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Appendix B

Overall Flow Evaluations

Appendix C

A subset listing of projects at which whitewater boating has been analyzed.

I. Introduction

Whitewater boating is a flow dependent recreational use of rivers, and considerable work evaluating flow-recreation relationships has occurred over the last several decades (Brown et al., 1991; Shelby, Brown, & Taylor, 1992; Whittaker and Shelby, 2002). Many of the flow-recreation studies focus on whitewater boating, such as rafting, kayaking, and canoeing, as flow often determines whether people have opportunities to take a trip and what level of challenge or social value is provided (Whittaker & Shelby, 1993). Different flow levels provide for varied whitewater boating opportunities. As flows increase from zero, different paddling opportunities and challenges exist within ranges of flows on a spectrum: too low, minimal acceptable, technical, optimal, high challenge, and too high. Standard methodologies are used to define these flow ranges based on individual and group flow-evaluations. The various opportunities provided by different flow ranges are described as occurring in "niches" (Shelby et al., 1997).

Whitewater Boating is enjoyed in different crafts, such as canoes, kayaks, and rafts. Different craft types provide different opportunities for river-based recreation, from individual or small group trips, to large group multi-day excursions. Flows that provide greater social value for one type of craft, such as canoes, may not provide equivalent social value for rafting. Changes in streamflow can have direct effects on the quality of whitewater boating, for every craft type. Direct effects may change quickly as flows change, such as safety in running rapids, number of boat groundings, travel times, quality of rapids, and beach and camp access (Brown, Taylor, & Shelby, 1991; Whittaker et al., 1993; Whittaker & Shelby, 2002). Indirectly, flows effect wildlife viewing, scenery, fish habitat, and riparian vegetation over the long term as a result of changes in flow regime (Bovee, 1996; Richter et al., 1997; Jackson & Beschta, 1992; Hill et al., 1991).

Streamflow is often manipulated through controlled reservoir releases, unanticipated spills from dams, and in-channel diversions. Additional scenarios, such as climate change and drought, water rights development, or conservation and the associated decreases in water demands, can all impact flows and recreation quality. Decision-makers within land and resource management and regulatory agencies, are increasingly interested in assessing the impacts of flow regimes on recreation resources. This has been most notable in the Federal Energy Regulatory Commission's (FERC) relicensing process, and where decision-makers, resource managers, and interest groups consider the extent that flow regimes can be managed to provide desirable recreational resource conditions. Appendix C lists a subset of projects where Whitewater Boating Flows have been analyzed. In these decision-making settings, specific evaluative information on how flow affects recreation quality is critical, particularly where social values are often central to decision-making (Kennedy and Thomas 1995).

Researchers collecting and organizing evaluative information often employ a normative approach using survey-based techniques. This approach is particularly useful for developing thresholds, or standards, that define low, acceptable, and optimal resource conditions for whitewater boating. Thresholds are crucial elements in any effective management or decision-making process (Shelby et al. 1992). The approach examines individuals' evaluations of a range of conditions (personal norms). Social Norms, defined by aggregate personal norms, describe a group's collective evaluation of resource conditions. This approach has been used to understand streamflows for whitewater boating on the Grand Canyon (Shelby et al. 1992), as well as several others rivers in Colorado (Vandas et al. 1990, Shelby & Whittaker 1995, Fey & Stafford 2009).

American Whitewater designed and conducted this study to collect evaluative information on whitewater boating attributes for 16 recreational attributes in the Yampa and White River Basins. Using overall flow-evaluation data, we developed flow-evaluation curves that graphically illustrate low, acceptable, and optimum flows for whitewater boating. In addition, specific flow evaluations were collected to aid in "calibrating" points along each curve. The present paper integrates both types of information in order to assist the Yampa-White River Basin Roundtable and the Colorado Water Conservation Board, in the defining non-consumptive flow-needs for recreation, and in the development of quantitative metrics that can be used to evaluate impacts from future water supply scenarios.

II. Recreational Flow Assessment – Locations and Methods

To define normative standards for whitewater boating flows in the Yampa-White River basin, American Whitewater collected and organized personal evaluations of recreational resource conditions, and recreation-relevant hydrology, consistent with NPS methodologies[2]. Using a web-based survey tool[3], American Whitewater designed two sets of questions asking respondents to evaluate flows for 16 river segments, relative to specific U.S. Geological Survey streamflow gage data.

Table A – Recreational Whitewater Attribute Locations

Reach Name	Streamflow Gage
Green River-Gates of Lodore	USGS - 09234500
Yampa River-Yampa Canyon	USGS - 09260050
Yampa River-Steamboat Town Run	USGS - 09239500
Yampa River-Lower Town Run (Steamboat Transit Center to Pump Station)	USGS - 09244490
Yampa River-Cross Mountain Gorge	USGS - 09251000
Yampa River-Little Yampa Canyon	USGS - 09247600
White River-Rangely to Bonanza Bridge	USGS - 09306290
Fish Creek	USGS - 09238900
Elk River-Box Canyon	USGS - 09241000
Willow Creek- CR 129 Bridge to Elk River Confluence	CDWR - WILBSLCO
Elk River-Box Canyon to Clark	USGS - 09241000
White River-above Kenny Reservoir	USGS - 09304800
White River - South Fork	USGS - 09304000
Mad Creek	Visual
Slater Creek	USGS - 09255000
Middle Fork Little Snake River	Visual

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[2] Whittaker, D., B. Shelby, J. Gangemi. 2005. Flows and Recreation, A guide to studies for river professionals. US Department of Interior, National Park Service, Anchorage, AK

[3] www.surveymonkey.com

The Flow-Evaluation Survey was based on the normative approach discussed in Section I, above. One set of survey questions was used to collect information that is used to develop overall flow-evaluations curves, and another set of questions helped identify and explain various points on those same curves. Overall Flow evaluation questions asked respondents to evaluate overall recreation quality for specific measured flows on each study segment, using a seven-point "acceptability" scale (unacceptable -3 and acceptable 3). This type of Survey contrasts with surveys that evaluate a single flow, or surveys conducted while flows are manipulated by controlled releases over a short period of time (Whittaker et al. 1993).

Another set of six specific flow evaluation questions asked respondents to report: 1) the minimum whitewater flow, 2) lowest preferred whitewater flow, 3) technical whitewater flow, 4) optimal whitewater flow, 5) high whitewater flow, and 6) highest safe whitewater flow. Respondents reported a single flow with respect to their preferred craft-type. A copy of the online Flow-Evaluation Survey, including both sets of questions, is attached as Appendix A.

An announcement of the flow-evaluation study was sent to over 5,000 American Whitewater members, including a link to the online survey website. The announcement was also posted to several online river-related discussion forums and various regional paddling club websites. The online format allowed whitewater boaters of all skill-levels and craft-types to report personal evaluations. The survey sample included outfitters currently permitted to operate commercially on targeted rivers, and non-commercial boaters. Because there were few differences between these groups, the data was combined in the analysis.

In all, 292 volunteer paddlers responded to the survey, although very few respondents had experience with every segment in the study. Table B summarizes the number of survey responses for each study segment. For this study, 81% of respondents identified themselves as private paddlers, 78% of respondents identified themselves as advanced or expert paddlers, and 43% reported paddling more than 20 days per season. A wide-range of whitewater craft types was surveyed, with rafters (63%), kayakers (31%), canoeists (5%) all represented.

Most respondents (55%) reported living in Colorado, though paddlers from 26 states participated in the survey. 73% of respondents felt comfortable estimating flows in cfs (cubic feet per second) on targeted river segments, while 8% of respondents reported feeling "uncomfortable" or "somewhat uncomfortable" estimating flows for study segments. Not every study participant therefore provided a personal evaluation of flows for every segment included in the survey – resulting in a range of respondent numbers across segments. Table B Summarizes Respondent Numbers for each segment.

For most segments, information collected through the online survey provided sufficient data to proceed with analysis and organization of personal evaluations of flows for whitewater boating. However, respondent numbers were low (less than 10) for several smaller attributes in the Yampa-White River basin, and data was not sufficient to develop levels of agreement between personal evaluations of flows and overall recreation quality. For those segments where respondent numbers were less than nine and considerable disagreement between responses exists, development of flow-evaluation curves was not possible.

Table B:
Recreational Whitewater Attribute Locations and Respondent Numbers

Reach Name	Respondent Numbers
Fish Creek	50 responses
Yampa River-Steamboat Town Run	54 Responses
Yampa River-Lower Town Run (Steamboat Transit Center to Pump Station)	16 Responses
Mad Creek	7 Responses
Elk River-Box Canyon	26 Responses
Slater Creek	6 Responses*
Elk River-Box Canyon to Clark	18 Responses
Willow Creek- CR 129 Bridge to Elk River Confluence	12 Responses
Yampa River-Little Yampa Canyon	22 Responses
Middle Fork Little Snake River	9 Responses
Yampa River-Yampa Canyon	102 Responses
Yampa River-Cross Mountain Gorge	51 Responses
Green River-Gates of Lodore	93 Responses
White River - South Fork	6 Responses*
White River-above Kenny Reservoir	8 Responses*
White River-Rangely to Bonanza Bridge	16 Responses

* Indicates segments not included in development of Flow-Evaluations Curves and Flow Agreement Acceptability Index (FAAI) analysis

III. Results and Discussion

A. Overall Flow Evaluations

For each study segment, mean responses from the overall flow evaluation questions (Table D) were plotted for each flow level, and connected to create a curve. In most cases, the curves show inverted U shapes where low flows and high flows provide low quality recreation conditions, while medium flows provide more optimal conditions. Flow Acceptability Agreement Index determines respondent agreement regarding the acceptability of each specific flow level. Figure 1 illustrates the Flow-Evaluation Curve for Segment 1 - Fish Creek, and defines optimum flows, a range of tolerable flows, and minimum flows. Table C describes respondent agreement for flows. Appendix B contains Flow-Acceptability Curves and FAAI data for each Yampa-White River study segment.

Figure 1

Flow Acceptability Agreement Index Curve for Fish Creek
(Flows represented are flow levels at USGS FISH CR AT UPPER STA NR STEAMBOAT)

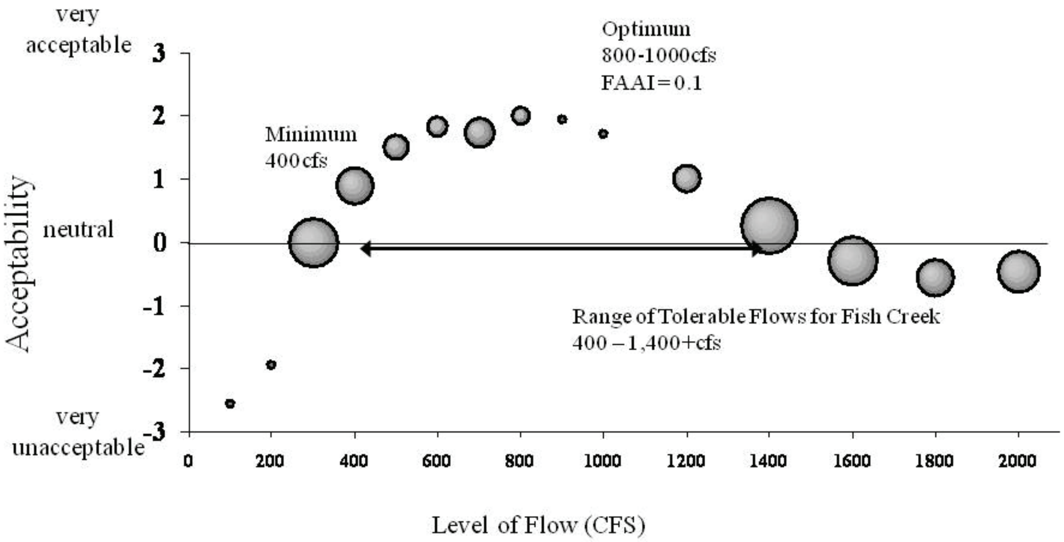


Table C

Flow Acceptability Agreement Index – Fish Creek
(Flows represented are flow levels at USGS FISH CR AT UPPER STA NR STEAMBOAT)

Specific Flow CFS	Mean Acceptability	FAAI
100	-2.55	0.03
200	-1.95	0.10
300	0	0.52
400	0.89	0.39
500	1.5	0.26
600	1.83	0.22
700	1.74	0.32
800	2	0.20
900	1.94	0.04
1000	1.71	0.04
1200	1	0.29
1400	0.25	0.58
1600	-0.29	0.52
1800	-0.57	0.38
2000	-0.47	0.44

The Flow Acceptability Agreement Index statistics show extremely high agreement levels for optimal flows (FAAI statistics range between 0 complete agreement, to 1 complete disagreement) while some level of disagreement between respondents exists in regard to the range of acceptable flows. The level of disagreement can be attributed to variations in flow-preferences between craft-types. Acceptable flows for kayaks may not provide equal value for rafts, for example. Additionally, personal skill or experience levels may impact overall agreement at the lower and higher end of the acceptable range of flows. Table E lists a subset of study segments, and the corresponding range of acceptable and optimal flows for both rafts and kayaks to illustrate the variability by craft-type.

Table D: *Acceptable and Optimal Flows for Whitewater Boating*

Whitewater Boating Attribute	**Minimum Flow (cfs)**	**Optimal Flows (cfs)**	**Acceptable Flows (cfs)**
Fish Creek	400	800-1000	400-1400
Yampa River-Steamboat Town Run	700	1500-2700	700-5000+
Yampa River-Lower Town Run (Steamboat Transit Center to Pump Station)	900	1500	400-4000
Mad Creek	400	400-1000	400-2000+
Elk River-Box Canyon	700	1000-2100	700-5000+
Slater Creek	600	1100-2100	600-3000+
Elk River-Box Canyon to Clark	700	1300-4000	700-5000+
Willow Creek- CR 129 Bridge to Elk River Confluence	300	700-800	300-1250
Yampa River-Little Yampa Canyon	1100	1700-2500	1100-10000+
Middle Fork Little Snake River	500	800-1100	500-2000+
Yampa River-Yampa Canyon	1300	2700-20000	1300-20000+
Yampa River-Cross Mountain Gorge	700	1500-3500	700-5000
Green River-Gates of Lodore	1100	1900-15000	1100-20000+
White River - South Fork	700	2500-3500	700-10000
White River-above Kenny Reservoir	700	1500-2500	700-10000+
White River-Rangely to Bonanza Bridge	700	1500-5000	700-10000+

Table E

Yampa-White Basin Segments
Minimum, Optimal and Acceptable Flows by Craft-Types

Yampa-White Basin Segment		Minimum Flow (CFS)	Optimal Flows (CFS)	Acceptable Flow (CFS)
Yampa – Steamboat Town Run	Raft	500	1100-2100	500-5000+
	Kayak	700	1100-2100	700-5000+
Yampa – Yampa Canyon	Raft	1500	3000-20000	1500-20000+
	Kayak	1100	1900-20000	1100-20000+
Green River – Gates of Lodore	Raft	1100	1900-10000	1100- 20000+
	Kayak	1100	1500-15000	1100-20000+

Utilizing Flow Acceptability Agreement Index and Flow-Evaluation curves, the range of acceptable and optimal flows for whitewater boating is defined for most segments. For three study reaches (Slater Creek, South Fork White River, and White River above Kenney Reservoir) response numbers were too low and did not provide sufficient data for curve development, though data was used to identify mean values for each flow range. For most other study segments, evaluations of higher flows never drop below the neutral line indicating that recreation quality may decline but may not drop below acceptable levels. For these segments, the high-end of acceptable flows listed in Table D, are not bound as indicated by the '+' symbol. In order to better understand the relationship between flows and recreation quality described by these Flow-Curves, study participants were presented a set of single-flow open response questions for each study segment.

B. Single Flow-Judgments

In order to further refine the overall flow-evaluation curves, a second set of single-flow evaluations were presented to survey respondents. For each study segment, survey respondents reported a single flow value that provides a distinct paddling experience or "niche" along a spectrum: minimum, low, technical, optimal, high challenge, and highest safe flow. To identify a single value for minimum flow, participants were asked "…what is the lowest flow required to navigate this stretch…" Alternatively, the Low Acceptable niche is differentiated from Minimum flow, as "the lowest flow you would return to [paddle] in your preferred craft, *NOT* the minimum flow that allows [navigation]. With single preference norms reported as specific flow values, measures of central tendency, such as the median, are useful representations of the flow in question. Median flow evaluations for each study segment are described in Table F.

Table F

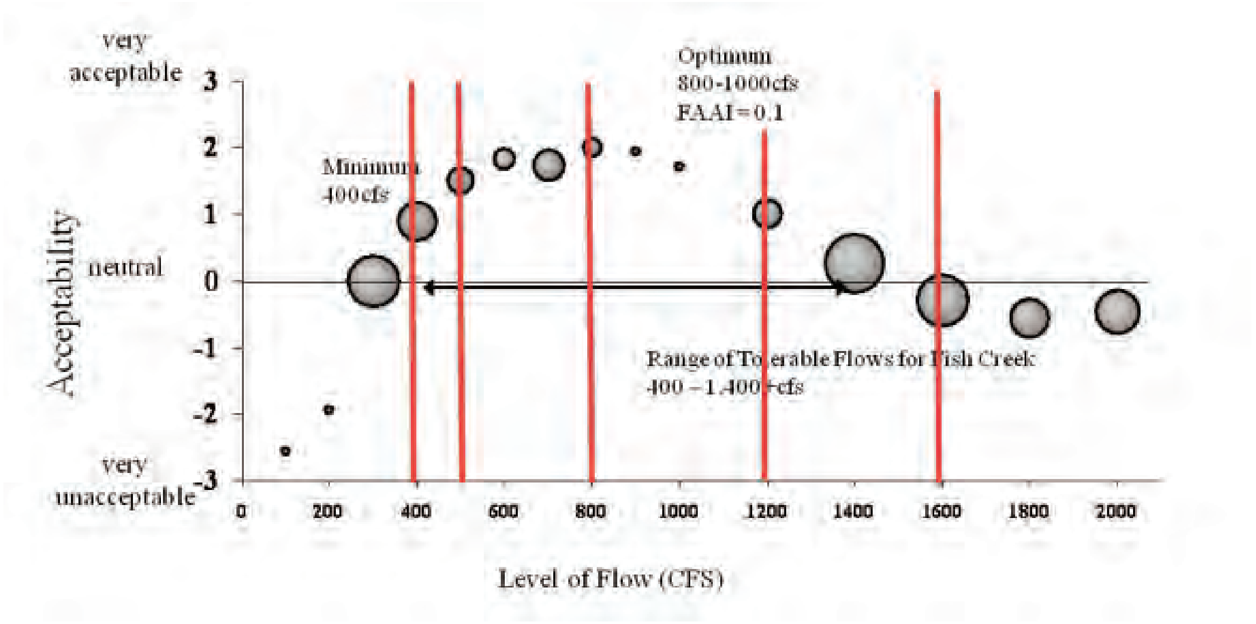
MEDIAN Minimum, Low, Technical, Optimal, High and Maximum Flows

Whitewater Boating Attribute	Minimum Flow (CFS)	Lowest Acceptable Flow (CFS)	Technical Flow (CFS)	Standard Flow (CFS)	High Challenge Flow (CFS)	Highest Safe Flow (CFS)
Fish Creek	400	500	500	800	1200	1600
Yampa River- Steamboat Town Run	300	600	700	1100	3000	4500
Yampa River- Lower Town Run (Transit Center to Pump Station)	400	500	900	1300	3500	5000
Mad Creek	300	400	500	500	800	1200
Elk River- Box Canyon	500	700	700	1000	3000	3000
Slater Creek	500	600	700	700	2000	2000
Elk River- Box Canyon to Clark	600	900	800	1200	3000	5000
Willow Creek	300	300	300	500	800	900
Yampa River- Little Yampa Canyon	800	1000	1000	2000	5000	10000
Middle Fork Little Snake River	400	500	500	800	1200	1600
Yampa River- Yampa Canyon	1000	1500	1500	5000	15000	20000
Yampa River- Cross Mountain Gorge	500	800	800	1700	4500	5000
Green River- Gates of Lodore	800	1000	1000	2000	8000	12000
White River - South Fork	500	600	600	1000	1500	2000
White River- above Kenny Reservoir	1000	1500	1500	5000	15000	20000
White River- Rangely	600	800	800	1000	3000	5000

C. Discussion

Overlaying the specific and overall flow-evaluation results is a helpful approach to analyzing the results of the study. An example of this integration is provided in Figure 2. Following along the FAAI curve, the median flow identified for minimum whitewater boating flows is 400 cfs (Table F), which is close to the point on the overall flow-evaluation curve where the neutral line between un-acceptable and acceptable valuation is crossed. Similarly, the median value for Standard or Optimal flows (800cfs) is close to the peak of the curve. Highest Safe flows (1600cfs) are close to the point where the FAAI curve drops below the neutral line, indicating that this flow provides low recreation quality. This approach to integrating results from both overall and specific flow-evaluation questions provides more information than either format by itself.

Figure 2
Integrating Single-Flow Evaluations and
Flow Acceptability Agreement Index Curve for Fish Creek



The results of this analysis show that good whitewater conditions (optimal flows) require higher flows, than those identified as providing minimum boatable flows. Good whitewater conditions for each target river segment have been identified in this study. For each study segment, the median response for minimum whitewater corresponds to the point where the overall flow-evaluation curve crosses above the neutral line. The median response for optimal flows however corresponds with the peak of the curve where ratings are highest. Overall Flow-evaluation curves are relatively flat at the top for most segments, which is attributed to the multiple tolerance norms captured in the study results. These Optimal flows are expressed as a range, in most cases.

IV. Conclusion

The purpose of the Flow-Evaluation Study conducted by American Whitewater, is to develop a baseline set of information that describe the relationship between streamflows and whitewater recreation in the Yampa•White Basin, such as rafting, kayaking, and canoeing. The study was based on two approaches to evaluating flows and recreation quality and includes personal evaluations of recreation quality and the structural norm approach, a technique used to graphically represent social norms. This approach has been utilized to identify flows needed to sustain the full range of whitewater boating opportunities on river stretches across the United States and Canada for over twenty years (Whittaker & Shelby, 2002). The graphic representation, commonly referred to as Flow-Evaluation or Impact Acceptability Curves, is used to describe optimum flows, ranges of tolerable flows, norm intensity and level of norm agreement (Shelby, Vaske, &, Donnelly, 1996). The Flow Agreement Acceptability Index (FAAI) takes the graphic representation of social norms one step further by displaying information about their central tendency, dispersion and form (Vaske, Needham, Newman, Manfredo, & Petchenik, in press).

For each of the river segments included in the analysis, high levels of agreement on optimal flows were recorded. Minimum acceptable flows were identified for each segment. For many segments, respondents reported no maximum acceptable flow; defining a wide range of acceptable flows exceeding 20,000 cfs for certain high volume runs. For most segments, single-flow judgments are shown to closely mimic relative values identified by the FAAI curves for minimum acceptable, optimal, and maximum acceptable flows. Median flow values for open-ended responses help describe specific flow-dependant “niches” for whitewater boating experiences along each FAAI curve.

Whitewater flow-preferences described in this summary report can be utilized in the future when evaluating future water management actions, climate change analyses, or risk management strategies. Based on the results of this flow-evaluation study, a usable days analysis can identify the average number of days in a given month that a river segment would be usable based on flow information. The purpose of the usable days analysis is to provide a baseline set of quantitative metrics to help evaluate the impacts to whitewater boating from future water management decisions in the basin.

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Appendix A:

American Whitewater's Online Flow-Evaluation Survey

Appendix B – Overall Flow Evaluation Results

Figure 1

Flow Acceptability Agreement Index Curve for Fish Creek
(Flows represented are flow levels at USGS FISH CR AT UPPER STA NR STEAMBOAT)

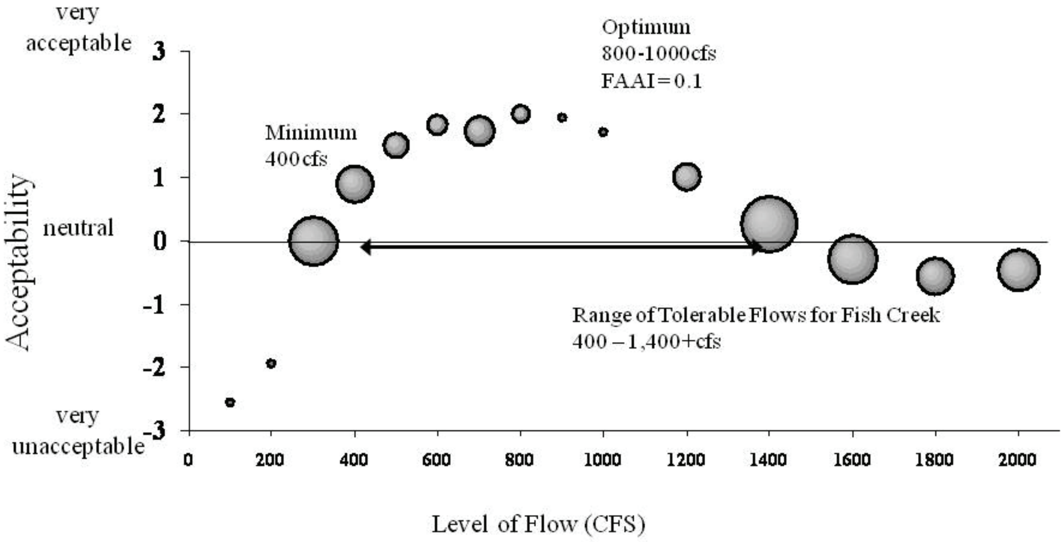

Table 1

Fish Creek Mean Acceptability Scores and
Flow Acceptability Agreement Index
(Flows represented are flow levels at USGS FISH CR AT UPPER STA NR STEAMBOAT)

Specific Flow CFS	Mean Acceptability	FAAI
100	-2.55	0.03
200	-1.95	0.10
300	0	0.52
400	0.89	0.39
500	1.5	0.26
600	1.83	0.22
700	1.74	0.32
800	2	0.20
900	1.94	0.04
1000	1.71	0.04
1200	1	0.29
1400	0.25	0.58
1600	-0.29	0.52
1800	-0.57	0.38
2000	-0.47	0.44

Figure 2

Flow Acceptability Agreement Index Curve for Steamboat Town Run
(Flows represented are flow levels at USGS YAMPA RIVER AT STEAMBOAT SPRINGS)

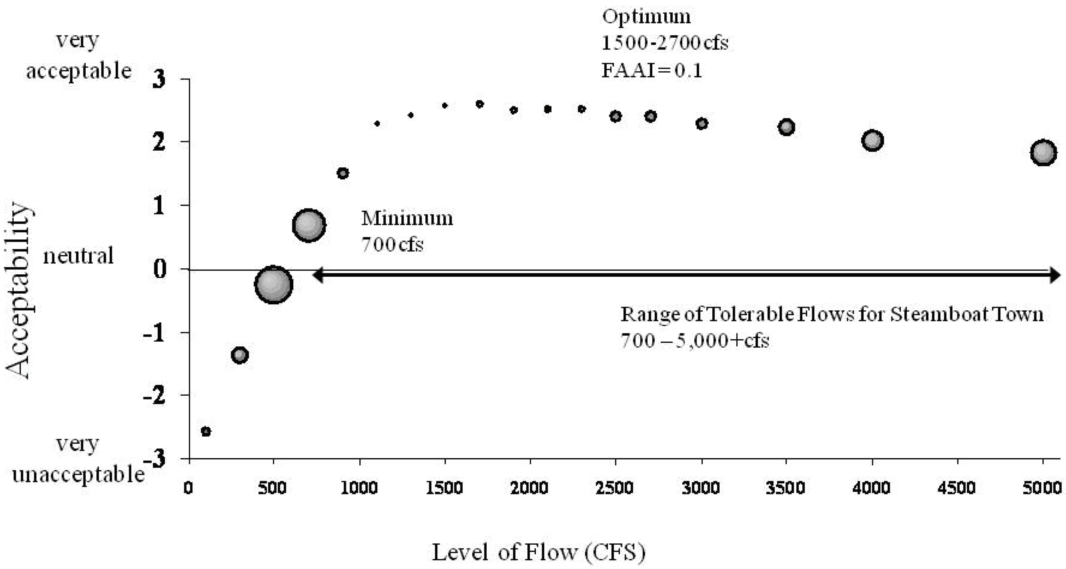

Table 2

Steamboat Town Run
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at USGS YAMPA RIVER AT STEAMBOAT SPRINGS)

Specific Flow CFS	Mean Acceptability	FAAI
100	-2.57	0.09
300	-1.37	0.17
500	-0.26	0.38
700	0.69	0.33
900	1.51	0.12
1100	2.29	0.04
1300	2.43	0.04
1500	2.58	0.04
1700	2.6	0.07
1900	2.5	0.08
2100	2.53	0.06
2300	2.52	0.06
2500	2.41	0.11
2700	2.4	0.11
3000	2.3	0.12
3500	2.23	0.16
4000	2.02	0.22
5000	1.84	0.27

Figure 3

Flow Acceptability Agreement Index Curve for Steamboat Town Run Rafts

(Flows represented are flow levels at USGS YAMPA RIVER AT STEAMBOAT SPRINGS)

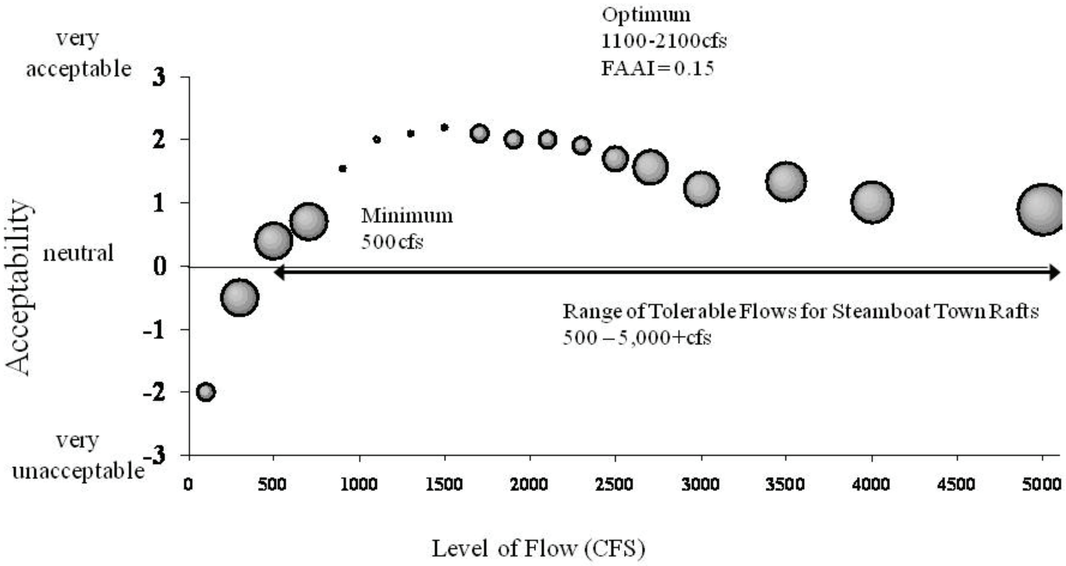

Table 3

Steamboat Town Run Rafts

Mean Acceptability Scores and Flow Acceptability Agreement Index

(Flows represented are flow levels at USGS YAMPA RIVER AT STEAMBOAT SPRINGS)

Specific Flow CFS	Mean Acceptability	FAAI
100	-2	0.20
300	-0.5	0.40
500	0.4	0.40
700	0.7	0.40
900	1.55	0.06
1100	2	0.07
1300	2.1	0.07
1500	2.2	0.07
1700	2.1	0.20
1900	2	0.20
2100	2	0.20
2300	1.9	0.20
2500	1.7	0.27
2700	1.56	0.37
3000	1.22	0.37
3500	1.33	0.41
4000	1	0.44
5000	0.89	0.56

Figure 4

Flow Acceptability Agreement Index Curve for Steamboat Town Run Rafts
(Flows represented are flow levels at USGS YAMPA RIVER AT STEAMBOAT SPRINGS)

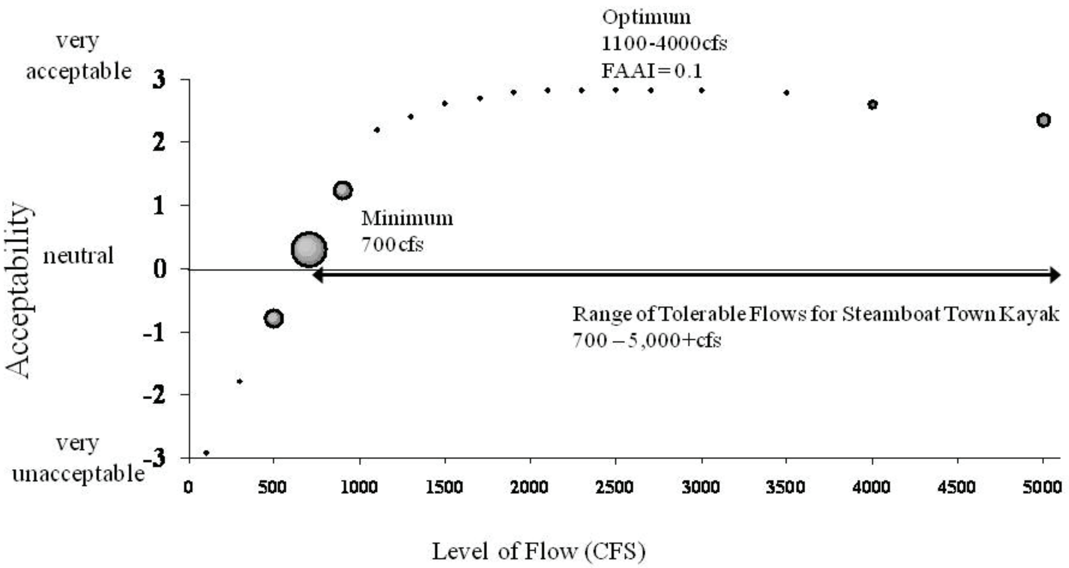

Table 4

Steamboat Town Run Rafts
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at USGS YAMPA RIVER AT STEAMBOAT SPRINGS)

Specific Flow CFS	Mean Acceptability	FAAI
100	-2.92	0.00
300	-1.79	0.02
500	-0.79	0.18
700	0.3	0.38
900	1.23	0.18
1100	2.2	0.04
1300	2.41	0.05
1500	2.62	0.05
1700	2.69	0.05
1900	2.79	0.05
2100	2.83	0.02
2300	2.83	0.02
2500	2.83	0.02
2700	2.83	0.00
3000	2.83	0.00
3500	2.79	0.01
4000	2.59	0.09
5000	2.34	0.15

Figure 5

Flow Acceptability Agreement Index Curve for Cross Mountain Gorge
(Flows represented are flow levels at USGS YAMPA RIVER NEAR MAYBELL, CO)

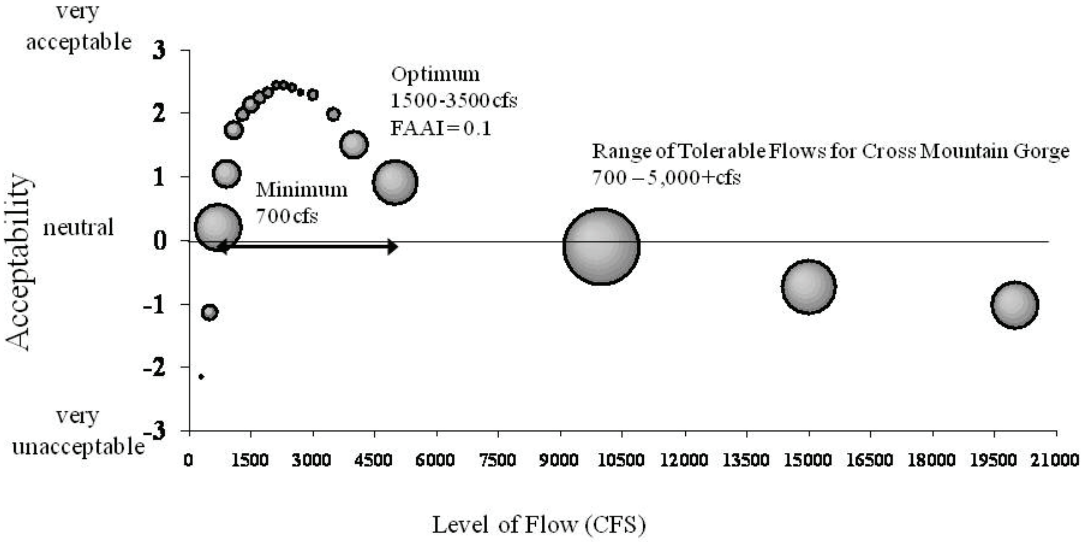

Table 5

Cross Mountain Gorge
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at USGS YAMPA RIVER NEAR MAYBELL, CO)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.16	0.05
500	-1.15	0.17
700	0.21	0.50
900	1.04	0.30
1100	1.74	0.18
1300	1.98	0.15
1500	2.14	0.17
1700	2.26	0.13
1900	2.32	0.12
2100	2.44	0.09
2300	2.45	0.09
2500	2.4	0.08
2700	2.33	0.06
3000	2.29	0.11
3500	1.98	0.15
4000	1.5	0.29
5000	0.92	0.46
10000	-0.1	0.79
15000	-0.73	0.56
20000	-1.03	0.49

Figure 6

Flow Acceptability Agreement Index Curve for Yampa Canyon

(Flows represented are flow levels at the USGS YAMPA RIVER AT DEERLODGE PARK, CO)

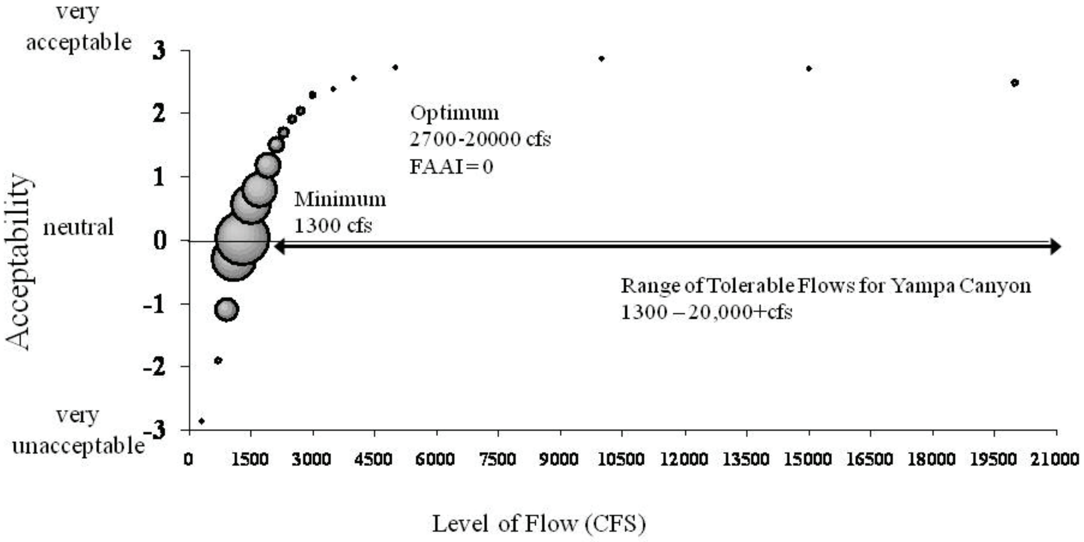

Table 6

Yampa Canyon

Mean Acceptability Scores and Flow Acceptability Agreement Index

(Flows represented are flow levels at the USGS YAMPA RIVER AT DEERLODGE PARK, CO)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.86	0.00
500	-2.51	0.01
700	-1.91	0.07
900	-1.11	0.24
1100	-0.3	0.46
1300	0.02	0.57
1500	0.57	0.41
1700	0.8	0.36
1900	1.17	0.27
2100	1.51	0.16
2300	1.69	0.12
2500	1.9	0.09
2700	2.05	0.08
3000	2.29	0.05
3500	2.39	0.05
4000	2.55	0.04
5000	2.74	0.03
10000	2.87	0.01
15000	2.71	0.04
20000	2.49	0.06

Figure 7

Flow Acceptability Agreement Index Curve for Yampa Canyon Rafts

(Flows represented are flow levels at the USGS YAMPA RIVER AT DEERLODGE PARK, CO)

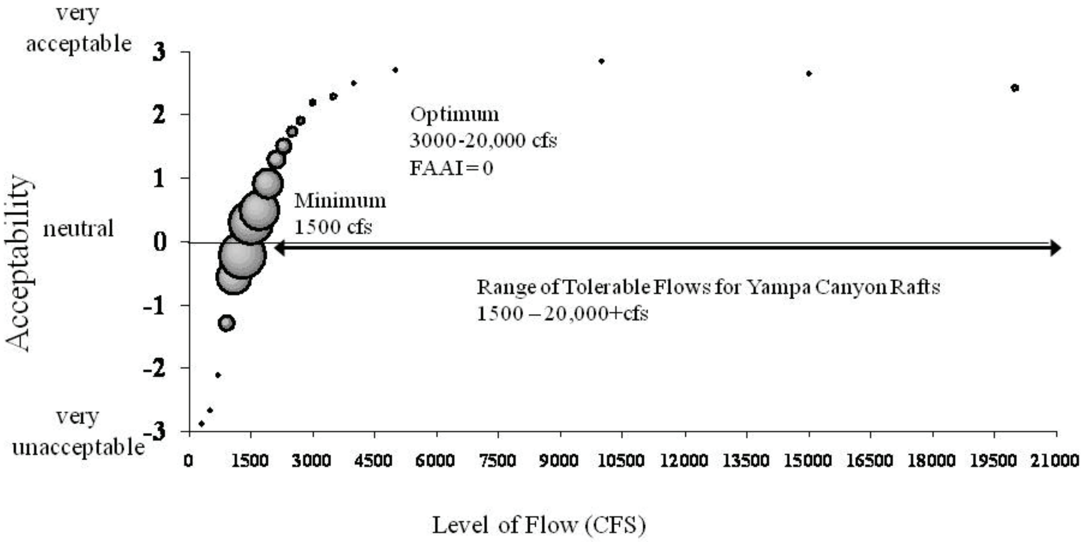

Table 7

Yampa Canyon Rafts

Mean Acceptability Scores and Flow Acceptability Agreement Index

(Flows represented are flow levels at the USGS YAMPA RIVER AT DEERLODGE PARK, CO)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.89	0.00
500	-2.67	0.00
700	-2.12	0.04
900	-1.29	0.18
1100	-0.57	0.36
1300	-0.23	0.48
1500	0.3	0.48
1700	0.49	0.42
1900	0.91	0.32
2100	1.3	0.20
2300	1.5	0.15
2500	1.74	0.11
2700	1.91	0.10
3000	2.19	0.07
3500	2.29	0.06
4000	2.5	0.04
5000	2.71	0.04
10000	2.85	0.01
15000	2.66	0.05
20000	2.42	0.07

Figure 8

Flow Acceptability Agreement Index Curve for Yampa Canyon Kayaks
(Flows represented are flow levels at the USGS YAMPA RIVER AT DEERLODGE PARK, CO)

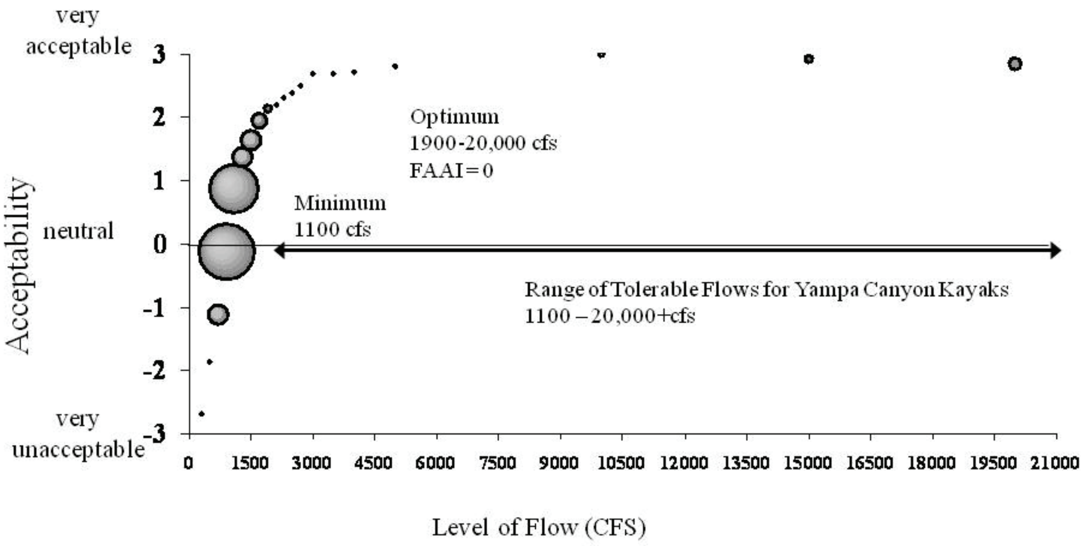

Table 8

Yampa Canyon Kayaks
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at the USGS YAMPA RIVER AT DEERLODGE PARK, CO)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.69	0.00
500	-1.87	0.04
700	-1.13	0.21
900	-0.13	0.58
1100	0.88	0.52
1300	1.38	0.21
1500	1.63	0.21
1700	1.94	0.17
1900	2.13	0.08
2100	2.19	0.04
2300	2.31	0.00
2500	2.38	0.00
2700	2.5	0.00
3000	2.69	0.00
3500	2.69	0.00
4000	2.71	0.02
5000	2.81	0.04
10000	3	0.07
15000	2.93	0.08
20000	2.85	0.13

Figure 9

Flow Acceptability Agreement Index Curve for Gates of Lodore
(Flows represented are flow levels at the USGS GREEN RIVER NEAR GREENDALE, UT)

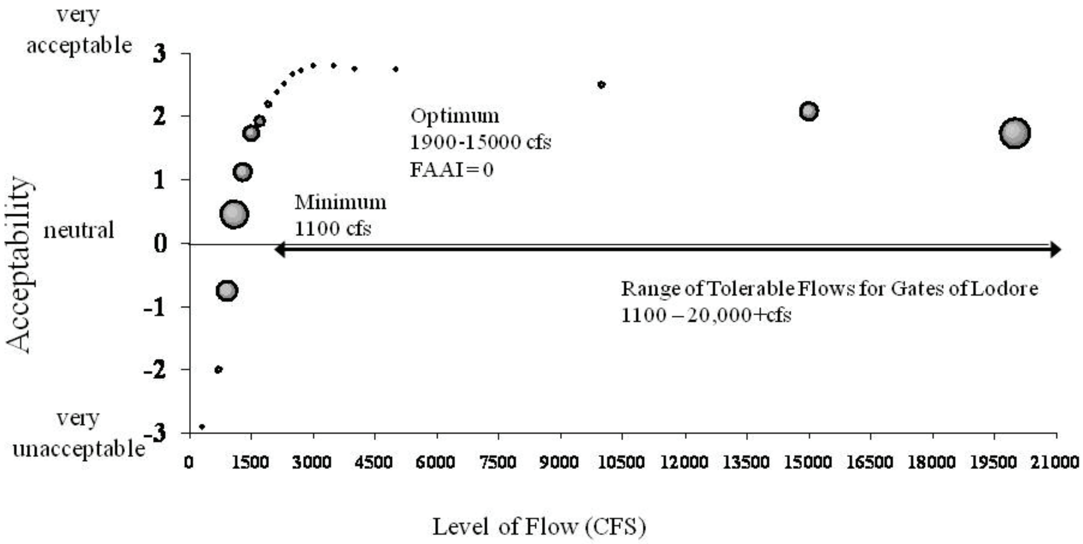

Table 9

Gates of Lodore
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at the USGS GREEN RIVER NEAR GREENDALE, UT)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.91	0.00
500	-2.65	0.01
700	-2	0.06
900	-0.75	0.20
1100	0.45	0.28
1300	1.12	0.20
1500	1.73	0.15
1700	1.93	0.11
1900	2.19	0.07
2100	2.39	0.04
2300	2.52	0.02
2500	2.67	0.01
2700	2.73	0.00
3000	2.8	0.00
3500	2.81	0.00
4000	2.76	0.02
5000	2.75	0.03
10000	2.51	0.06
15000	2.08	0.19
20000	1.74	0.31

Figure 10

Flow Acceptability Agreement Index Curve for Gates of Lodore Rafts
(Flows represented are flow levels at the USGS GREEN RIVER NEAR GREENDALE, UT)

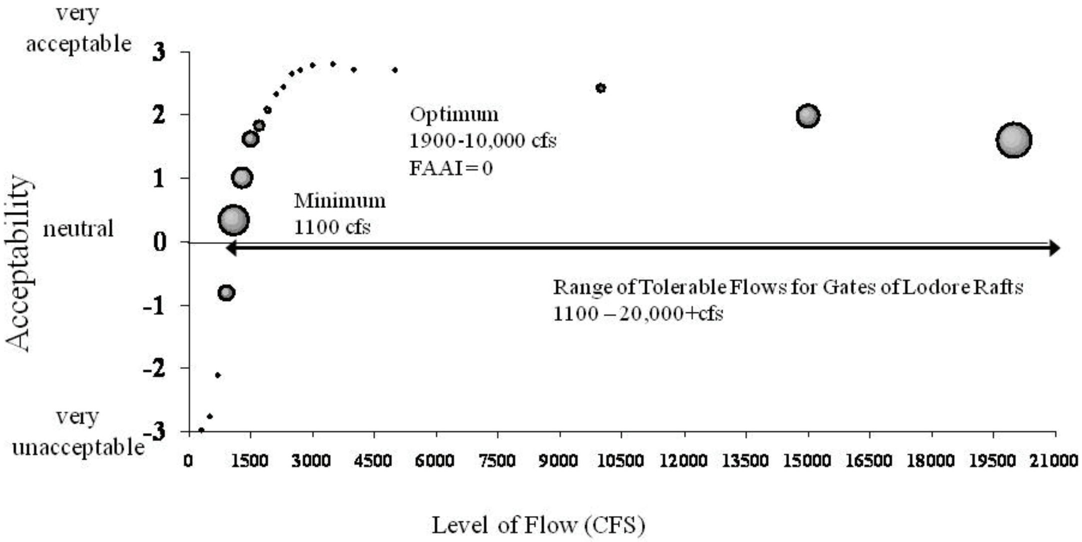

Table 10

Gates of Lodore Rafts
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at the USGS GREEN RIVER NEAR GREENDALE, UT)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.98	0.00
500	-2.77	0.00
700	-2.11	0.02
900	-0.81	0.17
1100	0.33	0.32
1300	1	0.22
1500	1.62	0.17
1700	1.83	0.10
1900	2.08	0.07
2100	2.33	0.03
2300	2.45	0.01
2500	2.65	0.00
2700	2.71	0.00
3000	2.79	0.00
3500	2.8	0.00
4000	2.72	0.03
5000	2.71	0.03
10000	2.43	0.08
15000	1.98	0.24
20000	1.6	0.37

Figure 11

Flow Acceptability Agreement Index Curve for Gates of Lodore Kayaks
(Flows represented are flow levels at the USGS GREEN RIVER NEAR GREENDALE, UT)

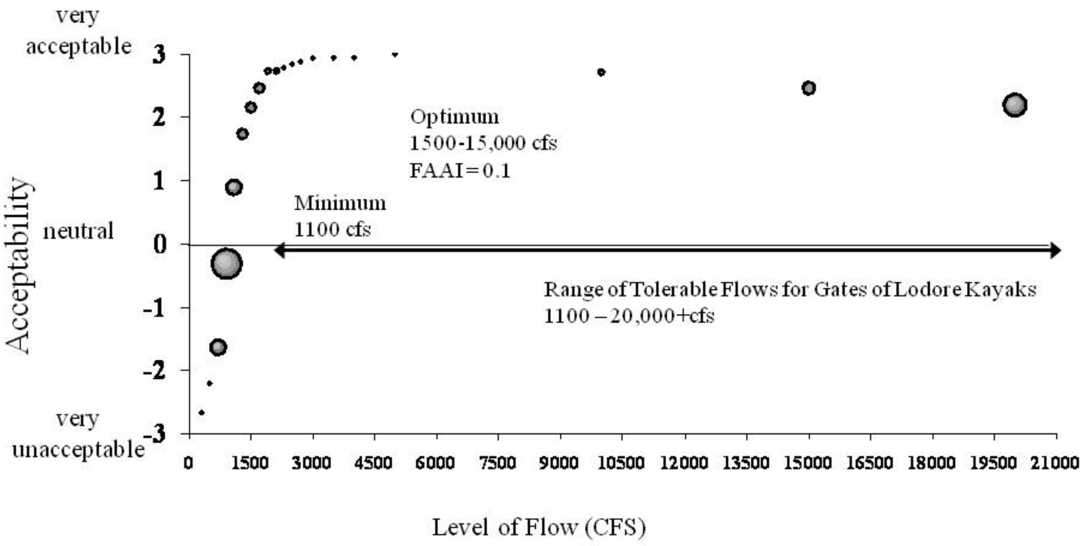

Table 11

Gates of Lodore Kayaks
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at the USGS GREEN RIVER NEAR GREENDALE, UT)

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.67	0.00
500	-2.22	0.04
700	-1.63	0.18
900	-0.32	0.32
1100	0.9	0.17
1300	1.74	0.11
1500	2.16	0.11
1700	2.47	0.11
1900	2.74	0.07
2100	2.74	0.07
2300	2.79	0.04
2500	2.84	0.00
2700	2.89	0.00
3000	2.95	0.00
3500	2.95	0.00
4000	2.95	0.02
5000	3	0.04
10000	2.71	0.06
15000	2.47	0.13
20000	2.2	0.24

Figure 12

Flow Acceptability Agreement Index Curve for Elk River Box
(Flows represented are flow levels at USGS ELK RIVER NEAR MILNER, CO*)*

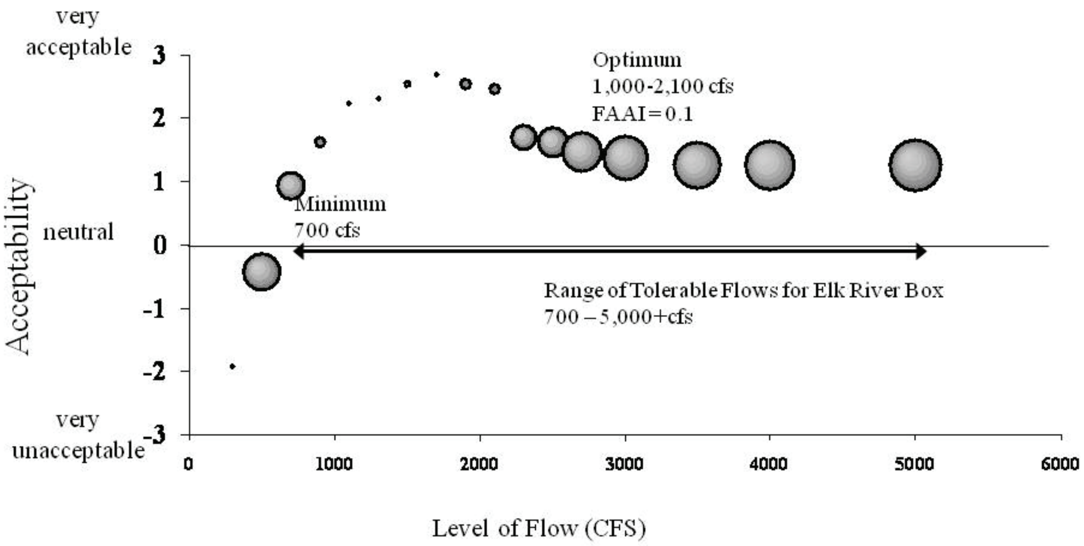

Table 12

Elk River Box
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at USGS ELK RIVER NEAR MILNER, CO*)*

Specific Flow CFS	Mean Acceptability	FAAI
300	-1.93	0.05
500	-0.43	0.38
700	0.93	0.29
900	1.62	0.10
1100	2.23	0.00
1300	2.31	0.00
1500	2.54	0.05
1700	2.69	0.05
1900	2.54	0.10
2100	2.46	0.10
2300	1.69	0.26
2500	1.62	0.31
2700	1.46	0.41
3000	1.38	0.46
3500	1.25	0.50
4000	1.25	0.53
5000	1.25	0.56

Figure 13

Flow Acceptability Agreement Index Curve for Elk River Below Box
(Flows represented are flow levels at USGS ELK RIVER NEAR MILNER, CO*)*

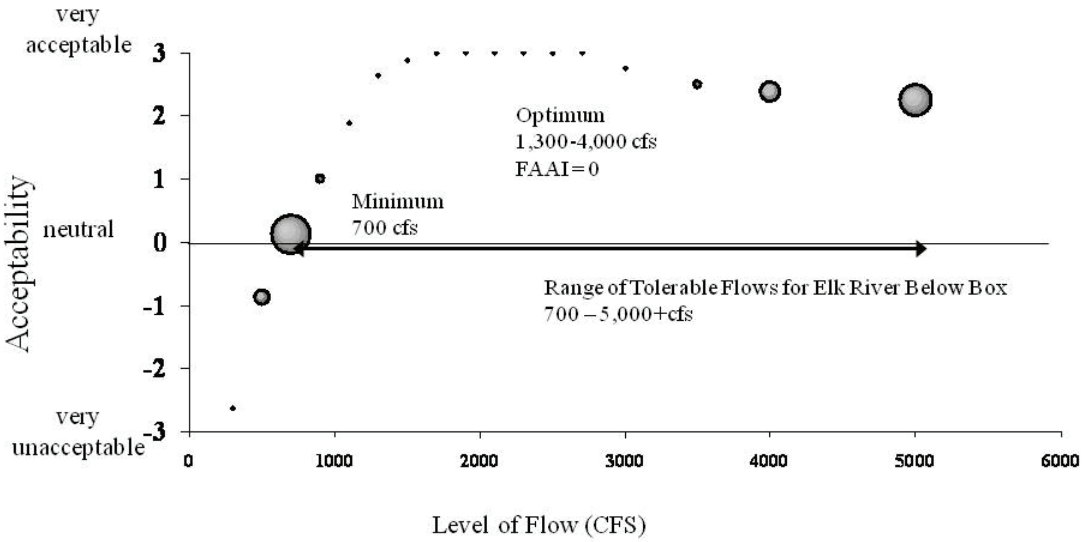

Table 13

Elk River Below Box
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at USGS ELK RIVER NEAR MILNER, CO*)*

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.63	0.00
500	-0.88	0.17
700	0.13	0.42
900	1	0.08
1100	1.88	0.00
1300	2.63	0.00
1500	2.88	0.00
1700	3	0.00
1900	3	0.00
2100	3	0.00
2300	3	0.00
2500	3	0.00
2700	3	0.00
3000	2.75	0.00
3500	2.5	0.08
4000	2.38	0.21
5000	2.25	0.33

Figure 14

Flow Acceptability Agreement Index Curve for Little Yampa Canyon
(Flows represented are flow levels at the USGS YAMPA RIVER BELOW CRAIG, CO*)*

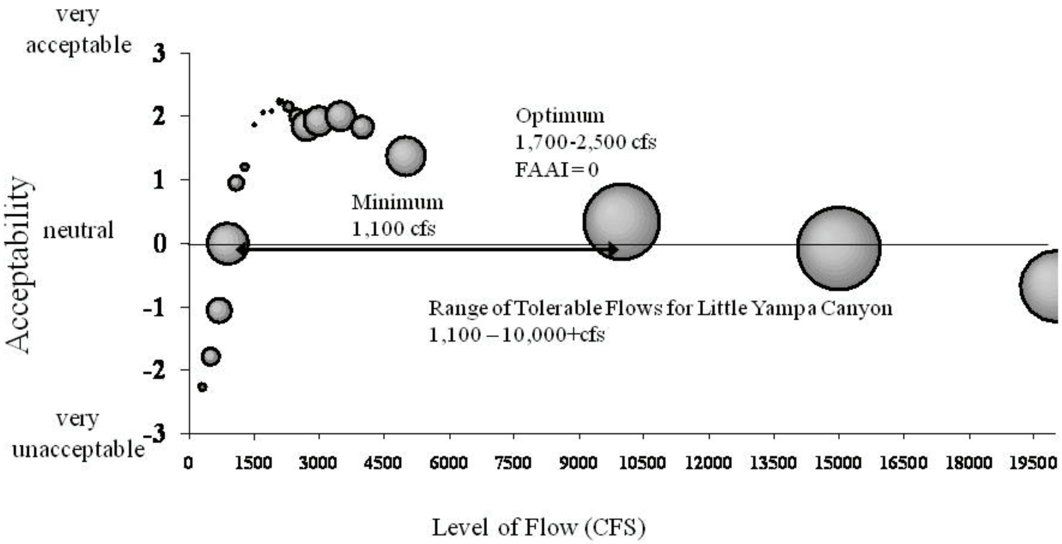

Table 14

Little Yampa Canyon
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented are flow levels at the USGS YAMPA RIVER BELOW CRAIG, CO*)*

Specific Flow CFS	Mean Acceptability	FAAI
300	-2.27	0.09
500	-1.8	0.18
700	-1.07	0.27
900	0	0.44
1100	0.94	0.17
1300	1.2	0.09
1500	1.87	0.00
1700	2.07	0.00
1900	2.08	0.00
2100	2.23	0.05
2300	2.15	0.10
2500	2	0.15
2700	1.85	0.31
3000	1.92	0.31
3500	2	0.31
4000	1.83	0.25
5000	1.38	0.41
10000	0.33	0.81
15000	-0.08	0.87
20000	-0.69	0.74

Figure 15

Flow Acceptability Agreement Index Curve for Mad Creek
(Flows represented are visual flow levels)

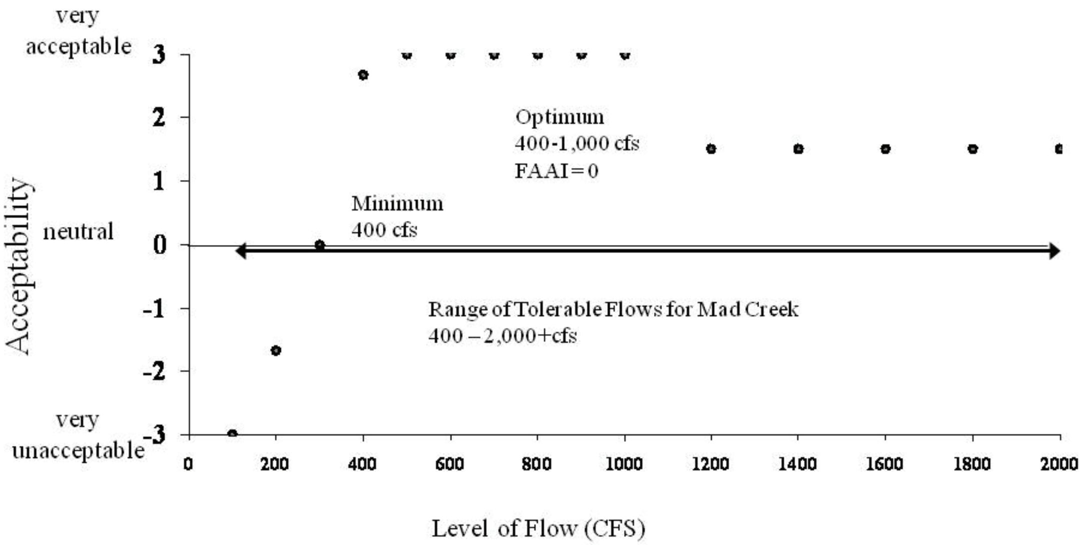

Table 15

Mad Creek Mean Acceptability Scores and
Flow Acceptability Agreement Index
(Flows represented are visual flow levels)

Specific Flow CFS	Mean Acceptability	FAAI
100	-3	0.00
200	-1.67	0.00
300	0	0.00
400	2.67	0.00
500	3	0.00
600	3	0.00
700	3	0.00
800	3	0.00
900	3	0.00
1000	3	0.00
1200	1.5	0.00
1400	1.5	0.00
1600	1.5	0.00
1800	1.5	0.00
2000	1.5	0.00

Figure 16

Flow Acceptability Agreement Index Curve for Yampa River
Transfer Center to Pump Station
(Flows represented at USGS YAMPA RIVER ABOVE ELKHEAD CREEK NEAR HAYDEN, CO*)*

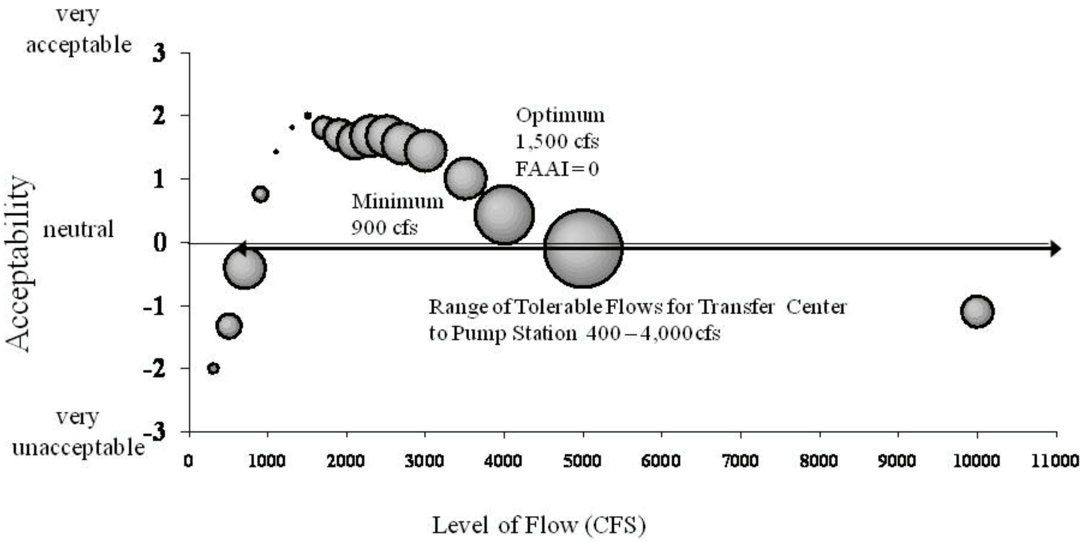

Table 16

Yampa River Transfer Center to Pump Station
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented at USGS YAMPA RIVER ABOVE ELKHEAD CREEK NEAR HAYDEN, CO*)*

Specific Flow CFS	Mean Acceptability	FAAI
300	-2	0.12
500	-1.33	0.28
700	-0.42	0.44
900	0.75	0.17
1100	1.42	0.00
1300	1.82	0.00
1500	2	0.06
1700	1.82	0.24
1900	1.7	0.33
2100	1.6	0.40
2300	1.67	0.44
2500	1.67	0.44
2700	1.56	0.44
3000	1.44	0.44
3500	1	0.44
4000	0.44	0.63
5000	-0.11	0.81
10000	-1.11	0.33

Figure 17

Flow Acceptability Agreement Index Curve for White River Rangely to Bonanza
(Flows represented at USGS WHITE RIVER BELOW BOISE CREEK, NEAR RANGELY, CO*)*

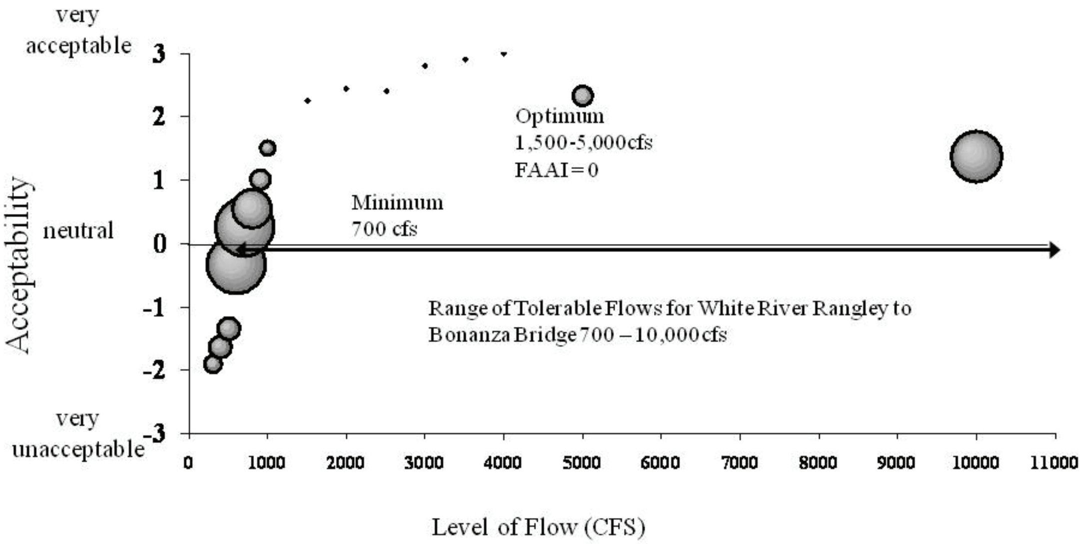

Table 17

White River Rangely to Bonanza
Mean Acceptability Scores and Flow Acceptability Agreement Index
(Flows represented at USGS WHITE RIVER BELOW BOISE CREEK, NEAR RANGELY, CO*)*

Specific Flow CFS	Mean Acceptability	FAAI
300	-1.91	0.18
400	-1.64	0.24
500	-1.36	0.24
600	-0.33	0.61
700	0.25	0.61
800	0.55	0.42
900	1	0.22
1000	1.5	0.17
1500	2.25	0.00
2000	2.45	0.00
2500	2.4	0.00
3000	2.8	0.00
3500	2.9	0.00
4000	3	0.00
5000	2.33	0.22
10000	1.38	0.54

Figure 18

Flow Acceptability Agreement Index Curve for Willow Creek

(Flows represented at USGS WILLOW CREEK, BELOW STEAMBOAT LAKE, CO*)*

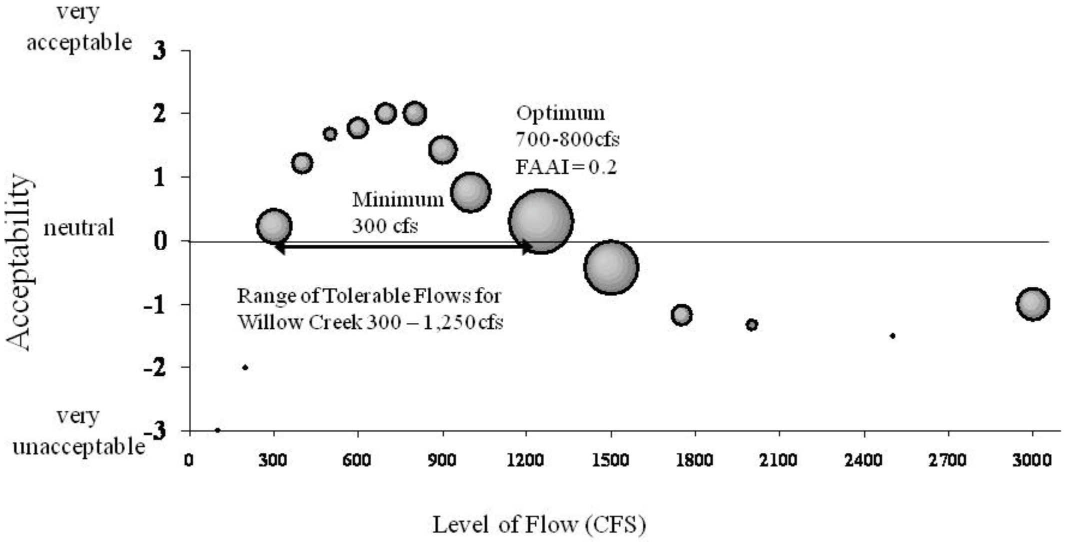

Table 18

Willow Creek

Mean Acceptability Scores and Flow Acceptability Agreement Index

(Flows represented at USGS WILLOW CREEK, BELOW STEAMBOAT LAKE, CO*)*

Specific Flow CFS	Mean Acceptability	FAAI
100	-3	0.00
200	-2	0.00
300	0.22	0.37
400	1.22	0.22
500	1.67	0.15
600	1.78	0.22
700	2	0.22
800	2	0.25
900	1.43	0.29
1000	0.75	0.42
1250	0.29	0.67
1500	-0.43	0.57
1750	-1.17	0.22
2000	-1.33	0.11
2500	-1.5	0.00
3000	-1	0.33

Appendix C

A subset of FERC regulated hydropower projects at which discrete usable boating days have been scheduled and/or provided as mitigation for impacts to whitewater boating, and/or analyzed as part of a whitewater flow study.

River	Project Name	State	FERC Project #
COOSA RIVER	JORDAN DAM	AL	00618
COOSA RIVER	MITCHELL	AL	00082
BUTTE CREEK	FORKS OF BUTTE	CA	06896
FEATHER RIVER	FEATHER RIVER	CA	02100
KERN RIVER	BOREL	CA	00382
KERN RIVER	ISABELLA	CA	08377
KERN RIVER	KERN CANYON	CA	00178
KERN RIVER	KERN RIVER NO 1	CA	01930
KERN RIVER	KERN RIVER NO 3	CA	02290
KINGS RIVER	PINE FLAT	CA	02741
MIDDLE FORK AMERICAN R	MIDDLE FORK AMERICAN RIVER	CA	02079
MIDDLE FORK STANISLAUS RIVER	BEARDSLEY/DONNELLS	CA	02005
N FK KINGS R	HAAS-KINGS RIVER	CA	01988
NORTH FORK FEATHER RIVER	POE	CA	02107
NORTH FORK FEATHER RIVER	ROCK CREEK-CRESTA	CA	01962
NORTH FORK FEATHER RIVER	UPPER NORTH FORK FEATHER RIVER	CA	02105
NORTH FORK MOKELUMNE RIVER	MOKELUMNE RIVER	CA	00137
PIRU CREEK	SANTA FELICIA	CA	02153
PIT RIVER	MCCLOUD-PIT	CA	02106
PIT RIVER	PIT 3, 4, & 5	CA	00233
PIT RIVER	PIT NO. 1	CA	02687
SAN JOAQUIN R	KERCKHOFF	CA	00096
SAN JOAQUIN RIVER	BIG CREEK NO 3	CA	00120
SAN JOAQUIN RIVER	BIG CREEK NO 4	CA	02017
SAN JOAQUIN RIVER	BIG CREEK NO.1 & NO.2	CA	02175
SOUTH FORK AMERICAN R	UPPER AMERICAN RIVER	CA	02101
SOUTH FORK AMERICAN RIVER	CHILI BAR	CA	02155
SOUTH FORK FEATHER RIVER	SOUTH FEATHER POWER	CA	02088
SOUTH FORK OF THE AMERICAN RIVER	EL DORADO	CA	00184
SOUTH YUBA RIVER	DRUM-SPAULDING	CA	02310
SOUTH YUBA RIVER	YUBA-BEAR	CA	02266
STANISLAUS R MIDDLE FORK	SAND BAR	CA	02975
STANISLAUS RIVER	SPRING GAP-STANISLAUS	CA	02130
WEST BRANCH FEATHER RIVER	DESABLA-CENTERVILLE	CA	00803
TALLULAH RIVER	NORTH GEORGIA	GA	02354

BEAR RIVER	BEAR RIVER	ID	00020
DEAD RIVER	FLAGSTAFF STORAGE	ME	02612
KENNEBEC RIVER	INDIAN POND	ME	02142
MAGALLOWAY RIVER	AZISCOHOS [?]	ME	04026
RAPID RIVER	UPPER & MIDDLE DAMS STORAGE	ME	11834
S BR PENOBSCOTT R	CANADA FALLS	ME	
W BR PENOBSCOT R	PENOBSCOT	ME	02458
W BR PENOBSCOT R	RIPOGENUS	ME	02572
SWAN RIVER	BIGFORK	MT	02652
WEST ROSEBUD CREEK	MYSTIC LAKE	MT	02301
PIGEON RIVER	WALTERS	NC	00432
TUCKASEGEE RIVER	DILLSBORO	NC	02602
WEST FORK TUCKASEGEE RIVER	WEST FORK	NC	02686
NANTAHALA RIVER	NANTAHALA	NC	02692
EF TUCKASEGEE	EAST FORK	NC	02698
ANDROSCOGGIN RIVER	PONTOOK	NH	02861
PEMIGEWASSET RIVER	AYERS ISLAND	NH	02456
HOOSIC RIVER	HOOSIC	NY	02616
MONGAUP RIVER	RIO	NY	09690
MOOSE RIVER	MOOSE RIVER	NY	04349
RAQUETTE RIVER	[STONE VALLEY REACH]	NY	
RAQUETTE RIVER	PIERCEFIELD	NY	07387
SACANDAGA RIVER	STEWARTS BRIDGE	NY	02047
SALMON R	SALMON RIVER	NY	11408
SARANAC RIVER	SARANAC RIVER	NY	02738
BEAVER RIVER	BEAVER FALLS	NY	02593
BEAVER RIVER	BEAVER RIVER	NY	02645
BLACK RIVER	GLEN PARK	NY	04796
BEAVER RIVER	LOWER BEAVER FALLS	NY	02823
BLACK RIVER	WATERTOWN	NY	02442
KLAMATH RIVER	KLAMATH	OR	02082
SOUTH FORK ROGUE RIVER	PROSPECT NO 3	OR	02337
SUSQUEHANNA RIVER	HOLTWOOD	PA	01881
SALUDA RIVER	SALUDA	SC	00516
WATEREE RIVER	CATAWBA-WATEREE	SC	02232
LITTLE TENNESSEE RIVER	TAPOCO	TN	02169
DEERFIELD RIVER	DEERFIELD RIVER	VT	02323
LITTLE RIVER	WATERBURY	VT	02090
LAKE CHELAN	LAKE CHELAN	WA	00637
SPOKANE RIVER	SPOKANE RIVER	WA	02545
SULLIVAN CREEK	SULLIVAN LAKE (STORAGE)	WA	02225
SULTAN RIVER	HENRY M JACKSON (SULTAN)	WA	02157
TIETON RIVER	TIETON DAM	WA	03701
BLACK RIVER	HATFIELD	WI	10805
CHIPPEWA RIVER	JIM FALLS	WI	02491