

Core Science Report For The Dolores River Dialogue (July 2005)



Dolores River Dialogue General Study Area Map

Acknowledgements

This report was the result of a collaborative effort by several scientists. These individuals are all committed to their individual areas of expertise and to the process of building a good foundation for the Dolores River Dialogue. This report was funded by The Nature Conservancy for the purpose of providing a summary of the work previously conducted on the lower Dolores River and provide insight into possible future areas of investigation.

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Executive Summary
For the
Dolores River Dialogue
Core Science Team Report
May 2005

The Dolores River Dialogue (“DRD or Dialogue”) is a multi-stakeholder effort focused on the environment of the Dolores River downstream of McPhee Dam. The dialogue is considering a range of creative alternatives. The practical actions that may result from this effort fall into three categories: 1) river channel work (maintenance, restoration, habitat improvement); 2) spill flow management / enhancement; 3) base flow – pool management / enhancement; and/or 4) some combination of the three. To evaluate the various strategies and determine the preferred alternatives, the Dialogue has convened a Core Science Team (CST) with technical expertise in several disciplines. This team was established in early February of 2005 to study and ultimately integrate four primary areas of investigation; Native warm water fisheries, Cold water fisheries, Geomorphology and Riparian Ecology with Hydrology through a community based stewardship approach.

The report presents the results of the CST’s initial investigations into conditions in the Dolores River. These evaluations include review of literature and previously prepared documents related to the Dolores River, discussion of linkages between flow and ecological and physical processes in the Dolores River, and identification of key data gaps. The information presented here provides a basis for future recommendations regarding flow management, channel work, and/or other management options.

Study Area:

The Dolores River flows from its headwaters in the San Juan Mountains to the confluence of the Colorado River in Utah. The study area consists of the Dolores River from McPhee Dam downstream to the confluence of the Colorado River, a length of approximately 200 river miles. The river courses through a range of natural communities and provides water for a number of municipal, industrial and agricultural uses. In order to provide a framework for analysis of conditions along the lower Dolores River and illuminate potential future management opportunities, Geographical Information Systems (GIS) were used to help define 8 reaches. These reaches were identified by differences in gradient, sinuosity, chemical parameters (e.g. salts) vegetative characteristics and potential limiting factors to natural stream channel movement and formation. In order to manage the science tasks, the river has been broken into two main focal areas. McPhee Dam downstream to Slickrock Bridge and Slickrock Bridge to the confluence with the Colorado River.

Core Science Individual Discipline Summaries

Geomorphology

The main conclusion of the Geomorphology report is that flows are the limiting factor to physical and ecological processes, and flow management is the key issue in terms of maintaining or restoring these processes. Flow is not only the single most important driving factor; it is also the driving factor that management has the greatest ability to influence. The analysis suggests that reductions in high flows below McPhee Dam are a primary cause of geomorphic changes in the Dolores River downstream of the dam. The primary changes that have occurred due to flow modification have likely included changes in channel dimensions, including narrowing and reduced depth, growth of lateral and mid-channel bars associated with reduced sediment mobility and encroachment of riparian vegetation. Other changes may be changes in the size, embeddedness, and mobility of bed sediments, simplification of bed morphology and reduced channel-floodplain connectivity.

Warm Water Fishery:

There are currently few rivers in the Colorado that have high biomass populations of the native fish species: roundtail chub, flannemouth sucker and bluehead sucker. Declining populations of roundtail chub and flannemouth sucker have been documented at both state-wide and basin-wide scales in recent years. The CDOW currently has fish monitoring sites established on the Dolores River which indicate roundtail chub and flannemouth sucker had a secure population in the Dolores River until recently. Habitat protection for roundtail chub in the Dolores River has become a priority issue for state fisheries managers and needs to be implemented quickly in order to avoid further population declines. Presumably the Dolores River genetic strain of roundtail chub, flannemouth sucker, and bluehead sucker is better fit to persist in the Dolores River, since they were able to persist given historic (gage records) flow conditions. Maintenance of a viable native fish community in the Dolores River potentially helps avoid additional native species being listed for federal protection.

Cold Water Fishery:

Reach 1, at least from McPhee Dam to Bradfield Bridge, was intensively evaluated in the late 1980's and early 1990's pertaining to cold water fishery (trout) issues. Water temperature may be an issue from Lone Dome to Bradfield Bridge. Habitat improvements would be beneficial and should be focused on enhancing physical habitats for juvenile and adult trout, not for spawning or rearing habitats. Early survey work conducted this year (1995) indicated a poor production of Rainbow Trout in Reach 1 and good production of Brown Trout in portions of Reach 2.

Riparian Ecology:

Most changes in community structure do not result from a single event or single stressor but from multiple interacting causes within a particular riparian corridor. Riparian vegetation plays a key role in the human-valued services provided by rivers and also provides the fundamental structure for diversity of flora and fauna found along the river.

A variety of factors influence the structure and composition of riparian vegetation growing along a particular reach of river, such as flows, salinity and soil type. Along the lower Dolores River study area a variety of habitats exist. These habitats range from a dominant Narrow leaf Cottonwood community in reach 1, Ponderosa Pine dominant in reach 2, Juniper and Pinion in

reach 3 and Tamarisk dominant in reaches 4-8. Willows are abundant throughout many of the reaches. However, Cottonwoods are not the dominant woody plant specie along the lower portion of the Dolores River and are found mainly in isolated patches with some of these patches demonstrating good age class distribution. Regulated flow of the Dolores River has greatly reduced volume discharge in reach 4 and may be one reason for the decline in Cottonwood regeneration. The solution to riparian habitat improvement may be found through the cooperative efforts of landowners, land managers, and ecologists.

Tamarisk:

Tamarisk is the dominant woody plant specie in many areas along the lower portion of the study area. Perennial flows and increases in salinity may be a contributing factor. Tamarisk may be having an influence on channel morphology by creating physical parameters that, during times of low flows, increase sediment deposition within in the channel. Tamarisk may also be limiting the normal recruitment and regeneration of native cottonwoods along the lower portion of the study area. Eradication of Tamarisk in “Key” areas may provide an opportunity for warm water fisheries habitat improvement.

Summary

(Science Coordinator)

The process of building a multi-disciplined report for the Dolores River Dialogue has provided an opportunity to bring new issues to the table concerning the science of the lower Dolores River. Although the Geomorphology report is adamant that flow is the number one reason for changes in the ecology of the river, other factors are also contributors to the process. McPhee Dam is now a part of the ecology of the river; therefore, looking for opportunities to enhance the ecology of the river with a combination of alternatives will likely result in greater success for all interested parties.

Core Science Document

Introduction

The Dolores River Dialogue (“DRD or Dialogue”) is a multi-stakeholder effort aimed at the environment of the Dolores River downstream of McPhee Dam. The dialogue is considering a range of creative alternatives. The practical actions that may result from this effort fall into three categories: 1) river channel work (maintenance, restoration, habitat improvement); 2) spill flow management / enhancement; 3) base flow – pool management / enhancement; and/or 4) some combination of these three strategies. Specific alternatives may include, but are not limited to, a review of the technical committee to instruct the Core Science Team to investigate the cost benefits of; re-timing downstream releases, efficiency/infrastructure improvements, interruptible supplies, new storage, new supplies, and stream habitat improvements.

To evaluate the various strategies and determine the preferred alternatives, the Dialogue has convened a Core Science Team with technical expertise in several disciplines.

This team was established in early February of this year (2005) to study and ultimately integrate four primary areas of investigation (Appendix A): Native warm water fisheries, Cold water fisheries, Geomorphology and Riparian Ecology with Hydrology through a community based stewardship approach. Within the Dolores River Dialogue plan to proceed the following provides the direction for the Core Science Team.

“PURPOSE”

“This Plan To Proceed outlines the three technical understandings required to get to the point where the Dolores River Dialogue Group can make a responsible decision about what, if any, action to take to implement its goals. First, a water availability analysis needs to be done. That analysis needs to describe the amount of water expected to flow downstream of McPhee Reservoir through spills and base flow releases. It also needs to describe the realistic opportunities to manage or enhance those flows. Second, an analysis of potential downstream environments needs to be made. The science associated with different flow patterns downstream of McPhee Reservoir needs to be described. Third, a correlation between those two efforts needs to be made that will illuminate the practical actions that could result from the efforts of the DRD Group. A matrix of doable alternatives with identified consequences (scientific, institutional, legal, political, and fiscal) will be described. The Plan’s finished products are designed to be thorough, credible, and realistic in their analysis of what is possible and what hurdles different actions may potentially face.”

The following tasks were included within the Plan to Proceed for the Core Science Team: identify flow options for the objectives described above to correspond to the climate scenarios (wet, normal, dry) described in the hydrology report; and be consistent with the available and

potential ‘opportunity’ flows described in the hydrology report. The report should identify criteria for adaptive management to be considered by the DRD.

The following report presents the results of the CST’s initial investigations into conditions in the Dolores River. These evaluations include review of literature and previously prepared documents related to the Dolores River, discussion of linkages between flow and ecological and physical processes in the Dolores River, and identification of key data gaps. The information presented here provides a basis for future recommendations regarding flow management, channel work, and/or other management options.

Study Area

The Dolores River flows from its headwaters in the San Juan Mountains to the confluence of the Colorado River in Utah. The river courses through a range of communities, from “alpine grasslands to montane forest areas to Sonoran desert lands”. The river provides water for a number of municipal, industrial and agricultural uses (Pontius, 1997).



Dolores River Dialogue General Study Area Map

Figure 1 (Image partially created using Delorme Topo USA)

In order to provide a framework for analysis of conditions along the lower Dolores River and illuminate potential future management opportunities, the study area has been divided into 8 reaches (Appendix B). These reaches were identified by distinct differences in gradient, sinuosity, chemical parameters (e.g. salts) vegetative characteristics and potential limiting factors to natural stream channel movement and formation.

Geographical Information Systems (GIS) were used to help define these differences within the study area. The study area consists of the Dolores River from McPhee Dam downstream to the confluence of the Colorado River, a length of approximately 200 river miles. Because this segment of river runs through both federal and private property the study and enhancement of potentially “Key” habitat areas along this segment of The Dolores River will require the cooperation of the both federal agencies and private property owners.

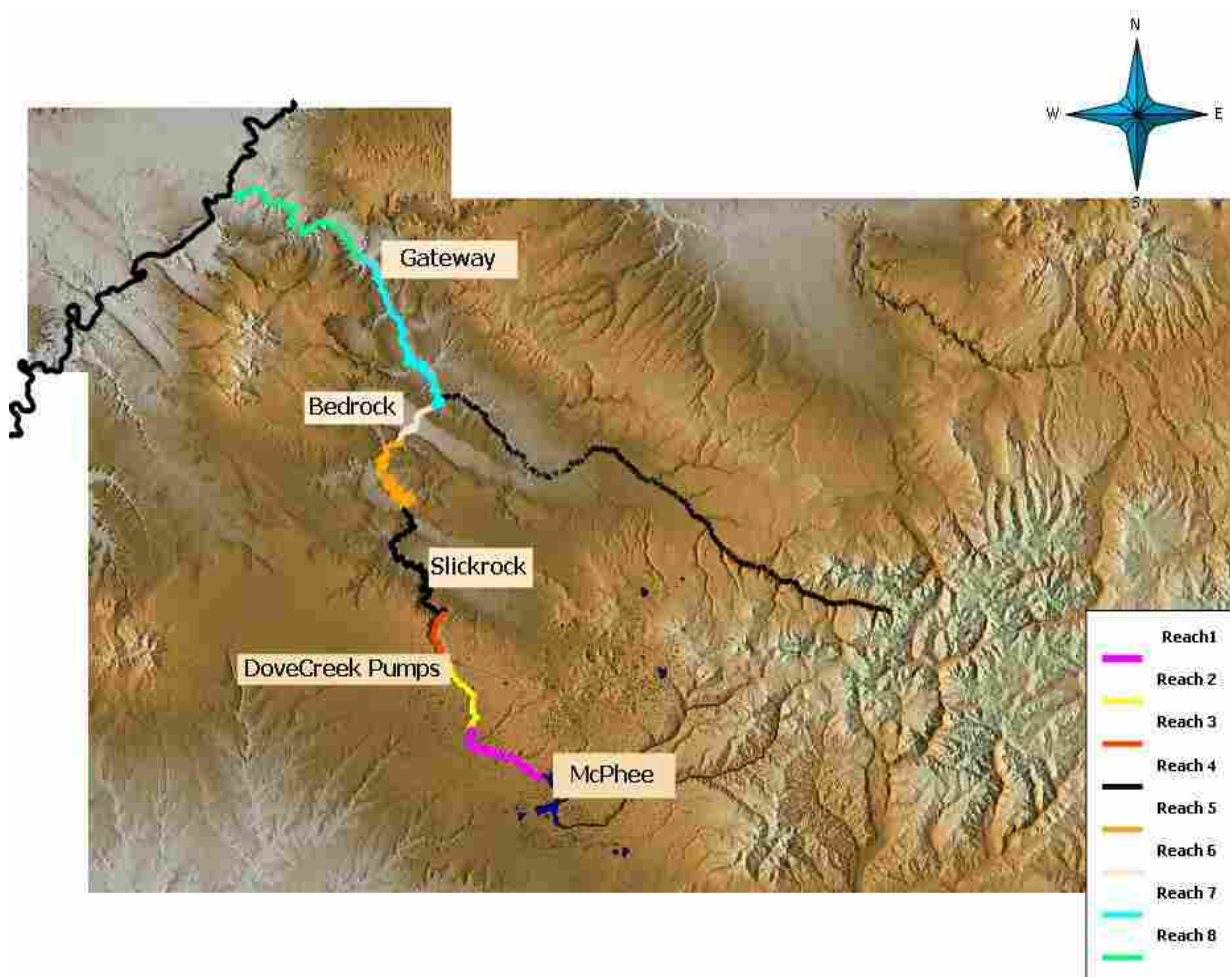


Figure 2 showing location of the 8 reaches

Dolores Project

McPhee Dam was completed in 1984 near the town of Dolores, Colorado, and impounds about 381,000 AF (229,000 AF of active storage) for irrigation, municipal and industrial water supply, hydroelectric generation, recreation, fish and wildlife enhancement, and flood control (U.S. Bureau of Reclamation 2001). McPhee Dam and Reservoir attenuate peak flows and augment base flows.

The Dolores Project was built to provide municipal and industrial water supply and supplemental and full-service irrigation water to the lands from the Dove Creek area in the north and south to the Ute Mountain Ute Tribe. The Project also stores water for the fishery downstream of McPhee Reservoir on the Dolores River. Prior to the Project, the Dolores River was often dry in late summer and fall. The fishery downstream of McPhee Dam was created by the Project as a result of the releases from the Fishery Pool.

Dolores River Watershed

The Dolores River watershed encompasses approximately 4,620 square miles in southwestern Colorado and southeastern Utah. Most of the lands within the watershed are owned by the U.S. Bureau of Land Management or the U.S. Forest Service.

Total discharge from the Dolores River into the Colorado River is approximately 544,000 af/yr (Pontius, 1997). Peak flows are highly variable with a 50-year high of 17,400 cfs (1958) and low of 1,260 cfs. The San Miguel River is the only significant tributary to the Dolores downstream from McPhee Dam (Roehm, 2001). Historic flow recommendations can be found in Appendix C.

Dolores River Watersheds			STREAM GAGE DATA					
Watershed Name	Size (mi ²)	% of Dolores R. Watershed	Watershed Name	USGS Gauge #	Elevation (ft.)	Period of Record	Watershed Size (mi ²)	Avg. Total Annual Q (AF)
WS1			Dolores River @ Dolores	09166500	6940	10/1/1895 - 9/30/1903; 10/1/1910 - 9/30/1912; 10/01/1921 - 9/30/2003 (92 yrs)	504	339,9381
Dolores River Headwaters	278.58		Dolores River @ Slickrock	09168730		5/1/1997 - 6/30/2003 (6 yrs)	1432	145397
West Dolores River	168.82		Disappointment Creek (near Dove Creek)	09168100	6420	8/1/1957 - 9/30/1986 (29 yrs)	147	19,4181
WS2			Dolores River @ Bedrock	09169500	4940	10/1/1917 - 9/30/1922; 8/1/1871 - 6/19/2003; (37 yrs)	2024	464,6091 280,8352 (340,526 / 227,186)
Lost Canyon Creek	194.83		San Miguel @ Ute	09177000	5000	8/1/1954 - 9/30/1962; 10/1/1973 - 9/30/2003; (38 yrs)	1499	262,2691
Plateau Creek	173.83		Dolores River @ Gateway	9179500	4548	10/1/1936 - 9/30/1954 (18 yrs)	4347	687208
Dolores R-Ponderosa Gorge	224.3		Dolores River @ Cisco	09180000	4165	12/1/1950 - 9/30/2003 (53 yrs)	4580	571,4932 (569,033 / 575,739)
WS3								
Disappointment Creek	345.3							
WS4								
Dolores R-Summit Canyon	185.98							
Dolores R-Gypsum Valley	163.62							
Dolores R-Paradox Creeks	132.34							
WS5								
San Miguel River	1555.23							
WS6								
Lower Dolores River	725							

Figure 3 (Data provided by CDOW)

Some have characterized our current understanding of stream ecology as at least “three integrated and interdependent components: the channel, riparian zone, and alluvial aquifer” Fischer, et al. (2001). One component of watershed management and integration of these interdependent components is the effects of land use practices on runoff, erosion, and off-site impacts. Land use practices and management can also have an effect on tributaries associated with watersheds. The management of the lands associated with these tributaries can influence the drainage networks associated with a watershed. Scott N. Miller, et al. (2000) state that these drainage networks play a critical role in routing water across and out of a watershed as well as the transport of sediment.

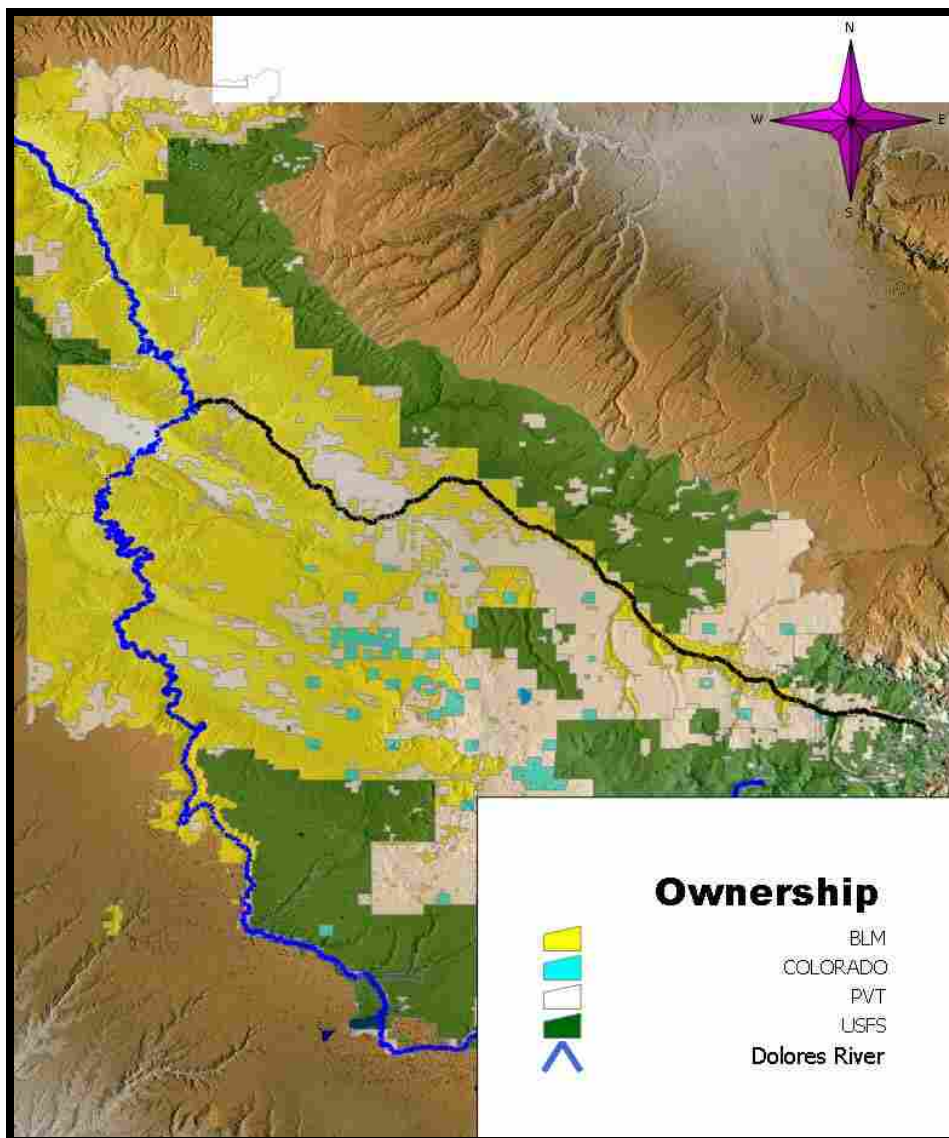


Figure 4 Map showing land ownership Dolores River Watershed

Many of these networks are impacted by the management of the watershed. Reid, (1993) describes these impacts as Cumulative watershed effects or (CWEs). He states “Cumulative watershed effects (CWEs) include any changes that involve watershed processes and are influenced by multiple land-use activities. CWEs do not represent a new type of impact, and almost all land-use impacts can be evaluated as CWEs. The CWE concept is important primarily because it identifies an approach to impact evaluation and mitigation that recognizes multiple influences. The significance of a CWE varies with the type of resource or value impacted and is determined on political, economic, and cultural grounds. In contrast, impact magnitude can be assessed objectively by measuring physical and biological changes. Most CWEs are incremental results of multiple controlling factors; rarely can a single threshold value be identified for provoking a response.”

Several HUC_10 watersheds (Figure 5) of the Dolores River System clearly demonstrate that at least two other factors (Salinity and Tamarisk) as well as hydrology are having an impact on the lower Dolores River. These factors in conjunction with other land use patterns may also be having an impact on sediment transport and input into the lower Dolores River. These factors can contribute to the formation and migration of the river channel and are likely to have a bearing on the available options for success within the lower Dolores River system.

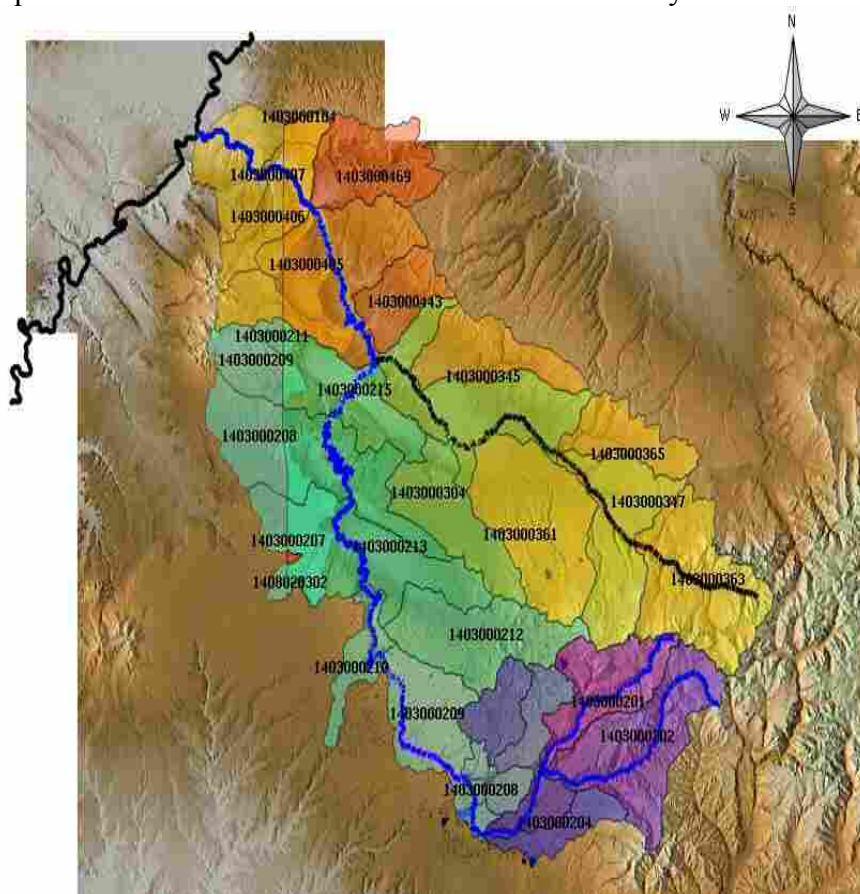


Figure 5 Map showing Huc_10 level watersheds for the Dolores River

Another important aspect of this study will be characterizing the riparian corridor along the entire study area. Fischer, et al. (2001), indicates that “Riparian definitions range from simple

descriptions, such as "associated with water courses"..., to technical and detailed descriptions for specific areas ". They cite a recent proposed definition that suggest riparian zones are, "three dimensional ecotones of interaction that include terrestrial and aquatic ecosystems, that extend down into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water, laterally into the terrestrial ecosystem, and along the water course at a variable width". . Fischer, Martin, and Fischenich (2000) define riparian zones as transitional areas between aquatic and upland terrestrial habitats and although not always well defined, they generally can be described as long strips of vegetation adjacent to streams, rivers, lakes, reservoirs, and other inland aquatic systems that affect or are affected by the presence of water. These zones typically comprise a small percentage of the landscape, yet they frequently harbor a disproportionately high number of wildlife species and "perform a disparate number of ecological functions when compared to most upland habitats" (Fischer, Martin, and Fischenich , 2000).

Throughout the southwest, it is well known how important riparian corridors are to water quality, and fisheries. Several factors or "ecological functions" can have very different effects on riparian ecosystems within a given reach or entire drainage basin. One of these "ecological functions" which can have a significant impact on cold water fisheries is temperature. Poole and Berman (2000), state that channel water temperature is determined by "the interactions between external drivers of stream temperature and the internal structure of the integrated stream system". They also state other factors can also affect stream temperature. The primary influences are climate "(such as air temperature and wind speed), stream morphology, groundwater influences, and riparian canopy condition". Other indicators of water quality include the study of macro invertebrates, which can serve as indicators of the general health of the river.

Elevation gradients will also have an effect of these factors. Gradients have been shown to show direct correlations between age class distribution of many riparian woody plant species. Cockman and Pieper, (1997). Land use and land cover changes may also have significant impacts on these systems. Different lands and cover types help predict "water chemistry aquatic invertebrate community composition and biotic integrity of fish", Gergel, et al., (1999).

Sometimes these challenges are also the result of surrounding uses of the land. Harding et al, (1998), clearly states this in historical context by saying, ecologists have "long recognized the strong dependence of streams on the surrounding terrestrial environment and these zones are influenced by stream hydrology, substrate characteristics, temperature regimes, and water chemistry, which in turn affect all trophic levels".

Summary

Based on the work completed by the Core Science Team, Core Hydrology team and concerns expressed by the Dolores River Dialogue during the course of this study, the following options may provide a useful strategy for success with the DRD;

- Develop Matrices of options for spill management using the recommendations from the Core Science Report.
- Develop options for spill management that improve the ability to manage for Quantity as well as Timing.
- Define “Key” habitat areas and needs for improving the roundtail chub populations.
- Improve riparian habitats in these “Key” areas by reducing Tamarisk abundance.
- Determine highest and best use for managing the “fishpool.”
- Continue to work within the “Community Based Stewardship” guidelines.

These options are intended to help manage the “fishpool” for the purpose of enhancing warm water fisheries habitat and to attempt to mimic historic flushing flows. In addition these options would attempt to coordinate the biological enhancement activities with recreational boating opportunities, and early runoff as well as flash flood events from tributaries along the lower Dolores River. These efforts would also need to coincide with the removal of tamarisk from “Key” habitat areas along the lower Dolores River.

The geomorphology report, portions of the riparian study, warm water fisheries and portions of the hydrology report all indicate that channel morphology, and warm water fisheries habitat has been degraded by the lack of water in the main channel of the Dolores River. However, within the entire 200 miles of the study area, gradient, vegetation and water quality, may also be having an impact on the ecology of the river. Therefore, the successful enhancement of the lower Dolores River ecosystem will require managing flows as well as work within the opportunities and constraints defined by the Dolores River Dialogue, as various options are discussed, in order to determine the costs and benefits to the system as a whole. The management of flows will also need to take full advantage of what appear to be cycles in the hydrograph. These cycles depending on duration and peak, can have substantial benefit for the Lower Dolores River.

This project will evolve as the current science effort is more fully integrated with the hydrology study. This integration will help policy makers manage flows, find areas important for habitat improvement and build tools to help manage both quantity of water and timing of runoff.

The following summaries of the individual disciplines outlined in the Plan to Proceed are intended to begin the process of integration of the Core Science with hydrology. These summaries are intended to stand alone as individual chapters in this report. Therefore, the

figures, tables and literature citations of each report are independent of one another, in terms of numerical order of Appendices and Figures.

Appendix G and H of the Core Science report are intended to provide background information on the habitat requirements for juvenile trout species and specific findings regarding the interactions between flow and temperature on Trout populations during winter in Reach 1 of the study area.

WARM WATER FISH LITERATURE REVIEW, by Rick Anderson (March 4, 2005)

The upper Colorado River basin, which is composed of the Colorado River and its tributaries upstream of Lake Powell, is home to 14 native fish species, four of which are now endangered. These four fish—the Colorado pikeminnow (*Ptychocheilus lucius*), razorback sucker (*Xyrauchen taxanus*), bonytail (*Gila elegans*) and humpback chub (*Gila cypha*) evolved in the Colorado River basin and exist nowhere else on earth (www.r6.fws.gov/coloradoriver). The Dolores River is a significant tributary to the Colorado River and thus the status of its native fish community is of keen interest to state and federal agencies that manage native fish.

The Colorado River Recovery Program commissioned a fish survey of the Dolores River downstream of McPhee Reservoir in 1990 and 1991 to determine this river's suitability for endangered Colorado River fishes (Valdez et al 1992). Prior to this investigation, little was known about historical or present use of the Dolores River by Colorado pikeminnow. Valdez et al. (1992) provided a review of earlier surveys:

‘The Dolores River once supported unknown numbers of Colorado pikeminnow (*Ptychocheilus lucius*) and perhaps functioned as a spawning tributary for this species in the upper Colorado River. Seethaler (1978) reported that T.M. Lynch seined small pikeminnow from Paradox Valley in 1962. Several other collections of Colorado pikeminnow were reported during the 1950's and 1960's by Lemons (1955), Nolting (1956) and Coon (1965). The most recent collection of Colorado pikeminnow in the drainage was an unconfirmed report by Horpestad (1973), who captured seven individuals in the San Miguel River, approximately 6 miles above its confluence with the Dolores River. No Colorado pikeminnow were captured during a fishery survey of the Dolores River by Holden and Stalnaker (1975) in 1972. More recent surveys by the U.S. Fish and Wildlife Service (Service) in the early 1980 are also failed to located Colorado pikeminnow in the Dolores River (Valdez et al 1982).’

Fish collections by Holden and Stalnaker (1975) included 11 species of which only four were native (flannelmouth sucker, bluehead sucker, roundtail chub and speckled dace). Valdez et al. (1982) collected a total of 16 species and also the same four native species. A total of 19 species, including six native and thirteen non-native fish were captured in the 1990 and 1991 by Valdez et al. (1992). The additional native fish collected in the 1991 survey were mottled sculpin and Colorado pikeminnow. Mottled sculpin were only found in Reach 6, a reach not sampled in prior surveys. Only four Colorado squawfish were collected in 1991 and all were located within 2 km of the confluence with the Colorado River. No other endangered species including bonytail, humpback chub or razorback sucker were captured by Valdez et al. (1992).

Valdez et al. (1992) employed a variety of sampling gear that included seines, gill nets, trammel nets and boat electrofishing. For all gear types combined the most common species captured were red shiner (33.4%), sand shiner (23.1%) and fathead minnow (18.4%), all non-native cyprinids. Combined the native species totaled 19% of the total catch with flannelmouth sucker

(9.2%) the most ubiquitous, followed by roundtail chub (4.6%), bluehead sucker (2.7%) speckled dace (2.5%), mottled sculpin (<0.1%) and Colorado pikeminnow (<0.1%) (Valdez et al.1992).

Valdez et al. (1992) found strong differences in native fish species composition between river reaches. These reaches were established to account for differences in water quality, geomorphology and flow conditions and are:

Reach 1: Dolores-Colorado River Confluence (RM 0.0 to the Utah-Colorado Stateline (RM 22.7).

Reach 2: Utah-Colorado Stateline to Salt Creek (RM 41.3).

Reach 3: Salt Creek to Dolores-San Miguel River confluence (RM 64.4)

Reach 4: Dolores-San Miguel River Confluence to Paradox Valley at Bedrock (RM 74.8)

Reach 5: Paradox valley at Bedrock to Dolores-Disappointment Creek Confluence (RM 128.7)

Reach 6: Dolores Disappointment Creek Confluence to Bradfield Bridge (RM 177).

Seining typically samples slow-shallow habitats dominated by smaller-bodied fish (length < 120 mm) like non-native cyprinids, speckled dace or age-0 of the large bodied native species. Red shiner, sand shiner and fathead minnow dominated seining collections in all habitats except riffles and isolated pools. Native species were only about 2% native fish collected in seines in Reach 1. Native composition by seining increased in an upstream direction and was about 10% in Reach 2 and 3, about 20% in Reach 4 and about 26% in Reach 5 (Valdez 1992).

Gill netting and boat electrofishing sample larger main channel habitats and deep pools. These gear types catch both smaller and larger sized fish and flannemouth sucker, roundtail chub, bluehead sucker, carp and channel catfish, were the most abundant species, respectively, and collected in 1990 and 1991 using these gear types.

In 1990 and 1991, flannemouth sucker were most prevalent in Reaches 3 and 4 at about 52 to 56 % of the catch with the netting and electrofishing gear. Flannemouth sucker were about 40 to 45% in Reaches 2 and 5, about 20% in Reach 6 and about 15% in reach 1.

Roundtail chub were common in the two upper Reaches (5 & 6) at about 30% of the netting and electrofishing catch. Roundtail chub were uncommon in lower Reaches comprising about 8% in Reach 4 and less than 5% in reaches 1, 2 and 3.

Bluehead sucker were more common in lower Reaches 1, 2 and 3 comprising from 11% to 18%, but were about 5% to 8% in upper Reaches 4, 5 and 6.

In 1990 and 1991, speckled dace were rare in lower Reaches 1, 2, 3, 4 and 5 with only 1 to 3% of the catch, but were common in reach 6 (23%).

Valdez et al (1992) reported no significant changes in species composition between the 1990 and 1991 samplings to a similar surveys done in 1981 and concluded the ichthyofaunal community remained relatively stable over that ten year period.

The Colorado Division of Wildlife has sampling locations established in the Dolores River. Mike Japhet (biologist in Durango, Colorado) has sampled a 1,000 ft reach below the Dove Creek pump station beginning in 1986 and has accumulated 16 years of data over the last 19 year period. Dove Creek site is located in Reach 6 defined by Valdez et al. (1992). The Dove Creek site was sampled by wade electrofishing using a stationary shore shocker. At least one pass were made, but in most years there were two passes at the site.

The three most common species collected at the Dove Creek site over the 18 year period were roundtail chub, speckled dace and mottled sculpin. Mottled sculpin was the most common species in 1986, 1987 and 1989 (about 50%), but ranked fourth in 2002 and 2003 and was fifth in 2004 at only 8%. Flannemouth sucker were rare at Dove Creek (1.3% for the period). Green sunfish were rare from 1986 to 2002 (0 to 1%), but ranked second in 2004 with 21% of the catch (CDOW data obtained from Mike Japhet).

The three most common species collected in 1991 in Reach 6 by Valdez were roundtail chub (27%), speckled dace (24%) and flannemouth sucker (22%). Mottled sculpin were the least common species (<1%) found by Valdez in 1991 in Reach 6.

Fish community composition was fairly stable at the Dove Creek site from 1986 to 2000 (Japhet, CDOW). During that time period the nonnative fish with the greatest influence were brown and rainbow trout. When trout were excluded, native species comprised between 95 to 100% of the remaining catch at Dove Creek site from 1986 to 2000. In 2002, 2003 and 2004 native composition dropped to 79, 76 and 67%, respectively (trout excluded). The large increase in fathead minnow and green sunfish after 2002 appear to be a result of habitat or temperature alternations due to reduced flow conditions in recent years.

Another Colorado Division of Wildlife study area was established by Rick Anderson (Researcher in the Grand Junction Office) in Big Gypsum Valley which is about 15 miles downstream of the confluence of Disappointment Creek in Reach 5. Anderson (2002) sampled the fish community at Big Gypsum in 2000 and 2001 as part of a habitat suitability/instream flow study. The Big Gypsum site was again sampled in 2004 as part of a drought evaluation study on various rivers (Anderson 2005).

Anderson (2002) did mark and recapture sampling that produced density and biomass estimates for fish larger than 150 mm. In 2000 the two most common species for larger fish (>150 mm) were roundtail chub (55%) and flannemouth sucker (16%) and these were also the most common fish in 2001 (Table 1). In 2004 roundtail chub were 29% and flannemouth sucker were rare at only 2%. The density estimates for roundtail chub and flannemouth sucker were very low on the Dolores River compared to the other study rivers and Anderson (2005) suggested a high degree of fine sedimentation had impacted productivity.

In 2004 native species comprised 53% of the total catch, less than in 2000 (81%) and 2001 (65%) (Table 1). Black bullhead was the most common fish > 150 mm (44%) in 2004, but was uncommon in 2000 (5%) and 2001 (0.6%). Anderson (2005) concluded that sedimentation appeared to have increased suitable habitat for black bullhead and he further speculated that predation by a large bullhead population could be responsible for the reduced flannemouth sucker and roundtail chub abundance in 2004. The Big Gypsum site is scheduled to be sampled in 2005 (Anderson 2005).

Table 1, Species composition for fish >150 mm and total fish in sample.

Species Composition	Fish over 150 mm			
	2000	2001	2004	2000
Flannemouth sucker	16	58	2	10
Bluehead sucker	2	6	1.2	1.0
Roundtail chub	55	25	39	51
Channel catfish	16	8	6	8
Carp	3	1.7	7	1.7
Green sunfish	2	1.4		4
Brown trout				0.3
Black bullhead	5	0.6	44	2.5
Speckled dace				18
Red shiner				2.8
Sand shiner				0.1
Fathead minnow				0.1
Native species	73.1%	87.9%	42.9%	80.5%

The shift in species composition for flannemouth sucker and roundtail chub between years (2000 and 2001) was similar to Reach 5 data reported by Valdez et al (1992) for collections made in 1990 and 1991. In 1990 the most common fish was roundtail chub (48%) followed by flannemouth sucker (35%), but in 1991 flannemouth sucker (51%) ranked first and roundtail chub (20%) second. Instability in species composition was not observed in other rivers with these native species (Anderson 2004). The most likely explanation is these species are dominated by young age-groups (Age 0 and Age 1) and minor changes in year class strength between years can result in large variations in abundance between years.

Native fish composition and abundance were found to be poor downstream of the San Miguel confluence, a river reach heavily impacted by poor water quality due to uranium tailings (Valdez et al. 1992). The river upstream of the San Miguel confluence appears to have the greatest potential for native fishes, but this river reach is highly vulnerable to altered flow regimes. Concerns about adequate instream flows and releases from McPhee dam have been an issue since the reservoir became operational in 1986. The Dolores River Biology team was formed in 1990 to provide biologically sound recommendations for managed releases from McPhee Dam (Mike Japhet, personal comm.). This Biology Team consists of one member from each of the following agencies and organizations: Bureau of Land Management, Bureau of Reclamation, Colorado Division of Wildlife, U.S. Fish and Wildlife Service, U.S. Forest Service, and Trout Unlimited.

The Biology Team considered their flow recommendations to be the minimum necessary to avoid degraded trout and native fish communities. As biological minima, they felt fish flows should not be subject to shared water shortages in dry years. In recent years releases from McPhee have been much less than the recommendations. The observed decline in native fish abundance appears to be a consequence reduced runoff and base flows. The provision that included the fish pool in sharing shortages during the recent drought period appears to be problematic for persistence of a thriving native fish community.

Currently management of roundtail chub, flannelmouth sucker and bluehead sucker has been a state and not a federal responsibility. There are currently few rivers in the state that have high biomass populations of these species and in recent years large declines have been documented in the Yampa River. Roundtail chub had a secure population in the Dolores River until recently. Habitat protection for roundtail chub in the Dolores River has become a priority issue for state fisheries managers.

WARM WATER FISH LITERATURE REVIEW, by Rick Anderson

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Geomorphology

Dolores River Dialogue Geomorphology Analysis

Gigi Richard and Andrew Wilcox

Introduction

Physical processes in rivers reflect a set of interactions between flow, sediment transport and channel morphology, and these in turn combine to form the physical template of aquatic habitat. River channel morphology reflects a long-term balance between a river's driving forces, including water discharge and channel gradient, and its resisting forces, including the size of sediment in the bed and banks, the frequency of bedrock exposures, bedform characteristics, riparian vegetation, channel curvature or sinuosity, and floodplain characteristics. Changes to a river's flow and/or sediment regimes, as may occur due to human uses of river systems, can alter these driving or resisting forces and produce adjustments in channel form and associated aquatic habitat. In particular, changes in river flows, sediment supply, or other controlling variables can alter channel dimensions, flow velocities, bed gradients, bed and bank sediment sizes and mobility, channel pattern (e.g., meander characteristics), and bedforms (e.g., distribution of pools and riffles). Because of the fundamental linkages between river morphology, aquatic habitat, and anthropogenic modification of river flow and sediment regimes, geomorphology, the study of landforms and the processes that shaped them, provides a valuable avenue for investigating the Dolores River downstream of McPhee Dam.

A preliminary assessment of geomorphology issues in the Dolores River downstream of McPhee Dam is presented below. This includes analysis of the following: (1) current and historical geomorphic conditions, (2) effects of flow modification on geomorphic processes, (3) sediment dynamics in the Dolores River, including both dam-related and non-dam related human effects; and (4) data gaps and future directions. This assessment is based primarily on previous reports, supplemented by limited analysis of aerial photographs and field observations, and is designed to provide information relevant to future management of the Dolores River. Further field investigations and historical analysis would be needed in order to develop a complete geomorphic assessment of channel conditions and changes compared to both pre-dam and presettlement conditions.

Geomorphic conditions in the Dolores River downstream of McPhee Dam

The defining characteristic of the Dolores River downstream of McPhee Dam is the series of steep-walled canyons, separated briefly by broad structural valleys, through which the river flows. Within these canyons, the river is flanked by steep sandstone cliffs and slopes with very shallow soils and narrow valleys and floodplains (CO DNR et al., 1976; USDA, 1972). Many segments of the canyon reaches of the Dolores River are likely bedrock-controlled, meaning that bedrock exposures in the channel bed and banks are common and that only relatively thin veneers of sediment would be present over the underlying bedrock over a longterm time scale. Another notable characteristic of the Dolores River is that because of the large elevation and

climatic range in the Dolores basin, geomorphic processes in the Dolores River have historically reflected a combination of both snowmelt-driven, montane hydrologic processes and arid, Colorado Plateau processes. Further descriptions of different portions of the Dolores River downstream of McPhee Dam are provided below.

McPhee Dam to Bradfield Bridge

Under pre-settlement conditions, this 11-mile reach was likely characterized by a channel that meandered along the entire valley bottom and bed morphology characterized by gravelcobble substrates and alternating pool and riffle sequences maintained by frequent (near-annual) pool scour and reworking of bars during spring high-flow periods. The floodplain flanking the channel likely supported significant riparian habitats of mixed deciduous forest comprised primarily of cottonwood, willow, and box elder (CDOW, 2003). The gradient of this reach is approximately 0.3% (~15 ft/mile), which is lower than the downstream canyon reach. Valley floor width in this reach is 0.25-0.5 miles (CO DNR et al., 1976). Following Anglo-European settlement, much of the floodplain forest was harvested for timber and converted to pasture; these pastures were irrigated by water diversions, and cattle were introduced (CDOW, 2003). Despite the negative effects of cattle grazing and diversion of the river's entire flow during lowflow periods on instream habitat conditions in this reach prior to construction of McPhee Dam, high flows continued to perform geomorphic work in this reach in the pre-dam era, maintaining channel and floodplain processes and riparian woodlands (Krieghauser and Somers, 2004). Since construction of the dam and reductions in high flows, however, geomorphic processes that support instream and riparian habitats, such as pool scour, bar formation, and floodplain rejuvenation have been largely absent from this reach, resulting in severely limited cover and spawning habitat and in embedded substrate conditions (CDOW, 2003), likely due to a combination of flow modification and historical impacts. In addition, a gravel road constructed along this reach has constrained the channel on the east side and limited channel migration. In response to the degraded conditions in this reach, the CDOW has implemented a habitat improvement project intended to establish a river morphology appropriate for the reach's modified flow regime. The project is designed to reduce channel width, restore channel sinuosity, increase the potential for overbank flooding, enhance riparian habitat, and improve instream trout habitat, including pool habitat and cover (CDOW 2003). The first phase of this project, implemented in 2003, entailed excavation of pools, construction of rock clusters and other structures designed to increase cover variability and promote pool scour, and importation of nearly 6000 yds³ of cobble to reduce channel width along a 1500-foot reach of channel in the Lone Dome State Wildlife Area (CDOW 2003).



Photograph of Dolores River in Lone Dome State Wildlife Area, between McPhee Dam and Bradfield Bridge, showing reach that has been narrowed through importation of cobble as part of CDOW habitat improvement project (from CDOW, 2003).

Bradfield Bridge to San Miguel River

Downstream of Bradfield Bridge, the river steepens and enters the Dolores Canyon, a deep canyon (up to 2300-ft. deep) with steep-sloped or stair-stepped canyon walls (CO DNR et al., 1976). The reach from Bradfield Bridge to Disappointment Creek (a 41-mile reach ending several miles upstream of the town of Slickrock) is the steepest reach downstream of McPhee Dam, with an average gradient of 0.45% (24 ft/mile) (CO DNR et al., 1976). Many segments of the river in Dolores Canyon are likely bedrock-controlled. Although the river has some sinuosity as it follows the entrenched meanders of the canyon walls, channel migration is limited due to the narrow valley bottom. A median grain size (D50) of 222 mm (large cobble) was measured near the Dove Creek pumping station in the late 1980s (USDI BLM, 1990). Below this location, a dirt road parallels the river for about 11 mile. This reach likely has a high sediment transport capacity because of its confinement and steep gradient. Downstream of Disappointment Creek, the river alternates between confined canyons and wide, structural valleys (CO DNR et al., 1976). This portion of the Dolores River is characterized by both reduced gradients and increased fine sediment loads (CO DNR et al., 1976), resulting in finer bed material than upstream reaches. For example, a median grain size of 60 mm (large gravel) was recorded near the confluence of Disappointment Creek in the late 1980s (USDI BLM, 1990); more recent data are not available. Sediment inputs from the Disappointment Creek drainage are particularly high and are discussed further below in the section on sediment dynamics. Below Disappointment Creek, after a short (6 mi.) canyon section, the Dolores meanders with a low gradient through Big Gypsum Valley, a broad valley that contributes minimal runoff to the mainstem. Bed substrates consist predominantly of siltsize materials, although the presence of coarser sediment along exposed bars suggests that fining may have occurred in this reach due to reduced flows and increased fine sediment supplies. From Big Gypsum Valley to Bedrock (a short distance upstream of the San Miguel River confluence), the river flows through Slickrock Canyon, a narrow canyon of highly sinuous, entrenched meanders flanked by steep sandstone cliffs, some

of which rise vertically over 1000 feet from the water's edge (CO DNR et al., 1976). The river's gradient in this reach is 0.21% (11 ft/mile) (CO DNR et al., 1976).

San Miguel River to Colorado River

The San Miguel River is the largest tributary to the mainstem Dolores, and because it is relatively unregulated flow volume increases substantially here at certain times of the year. Downstream of the San Miguel confluence, the Dolores River flows through another narrow deep canyon for 4 miles before the valley again widens, after which the river is paralleled by Highway 141 and numerous gravel and dirt roads. The 31-mile reach from the San Miguel confluence to Gateway has a gradient of 0.19-0.28% (10-15 ft/mile). Downstream of Gateway, the river initially meanders through a broad (approximately 2-mi. wide) valley flanked by a steep, sandstone dominated mesas rising 800 to 2500 ft. above the valley floor (River mile 31-17), before entering a narrow, steep-walled canyon (about 0.25-mile wide) past the state border (River mile 17-11) (USDI, 1979). Bed substrates consist of sand, gravel, and boulders derived from episodic debris flows out of tributaries, and from rockfall from canyon walls. The lowermost portion of the Dolores River (River mile 11-0) flows through soft sediments and a wider valley before entering the Colorado River (USDI, 1979).

Effects of flow on downstream geomorphic processes

The effects of diversions and storage on baseflows and high flows downstream of McPhee Dam are described in detail in the Dolores River Dialogue Hydrology Report. From a geomorphic perspective, the magnitude, frequency, and duration of high flows are the most important elements of a river's flow regime. The following section describes the geomorphic importance of high flows, historic changes in the geomorphically important components of flow regimes in the Dolores River, channel-forming discharge estimates, and the expected effects of flow modifications on river morphology.

The role of high flows in driving geomorphic processes

In recent years, river scientists have identified flow as a master variable controlling river ecosystem function and have developed a strong conceptual understanding of how natural flow regimes contribute to the health of river ecosystems (Poff et al. 1997, Bunn and Arthington 2002). Understanding of the geomorphic importance of high-flow components of river flow regimes is especially well established. Peak flows are important for formation and maintenance of the shape and form of the river channel (Leopold et al., 1964; Wolman and Leopold, 1957; Wolman and Miller, 1960). Increases in river discharge are accommodated by increases in the width, depth and/or velocity of the flow, creating greater forces on the channel bed and resulting in increased ability of the river to mobilize larger and larger sediment sizes. Mobilization and transport of bed sediment at high flows is responsible for maintaining channel form and can result in flushing of fine sediments, reworking of gravels, formation and maintenance of pool and riffle sequences, and formation and/or erosion of river bars. High flows play an important role in mobilizing and transporting coarse and fine sediment inputs delivered from tributaries. In addition, high discharges that flow out of the channel banks carry sediment that deposits on the floodplain, rejuvenating the floodplain surface and facilitating reproduction of certain riparian species. High flows also counteract the effects of vegetation encroachment and the growth of mid-channel bars by disturbing vegetation establishment within the active channel, including on mid-channel bars, and on floodplain surfaces. The geomorphic effectiveness of high flows is a

function of their magnitude, frequency, and duration. For example, the magnitude of high flows interacts with the existing channel and valley shape to determine the maximum width, depth and velocity of the flow, including the extent of floodplain inundation. High-flow magnitudes also determine the size of bed substrates that can be mobilized and whether bed sediments are transported as bedload (i.e., rolling or bouncing along the channel bed) or as suspended load (i.e., suspended in the water column). The frequency of high flows determines not only how often processes such as sediment transport, bank erosion, and vegetation scouring occur, but also temporal sequence of high-flow events and the amount of intervening time in which channel adjustments associated with low flows, such as encroachment of riparian vegetation or infiltration of fine sediment into the channel bed, can occur. High-flow duration influences the total amount of sediment transport and other geomorphic work performed by the high flows and can, by determining the extent of floodplain inundation, mediate effects on riparian vegetation. These high-flow components, as well as the timing and rate of change of high flows, are also extremely important to aquatic organisms (Poff et al., 1997).

Alterations to high flows in the Dolores River

Impacts to the flow regime of the Dolores River occurred before the construction of McPhee Dam. Prior to construction of McPhee Dam, the mean annual flow at Dolores (above McPhee Dam) was 763 cfs and decreased to 465 cfs at Bedrock (94 miles downstream from the dam). The decrease was caused by diversions in place prior to construction of McPhee Dam (BLM, 1990). McPhee Dam increased the depletion of the annual flows from 30% to 69% of natural flow (USDI BLM, 1990). Construction of McPhee Dam in 1984 affected the flow regime of the Dolores River by altering the spring peak flows and the magnitude and variability of the base flow. Between 1986 and 2004, the spring peak was essentially eliminated downstream from the dam for six of the 19 years of record (Figure 1 and Figure 2). In an average runoff year, both the magnitude and duration of the spring peak flow are decreased (Figure 3). Correlation of the peak flows above and below the dam show a distinct decrease in the peak flows below the dam (Figure 4).

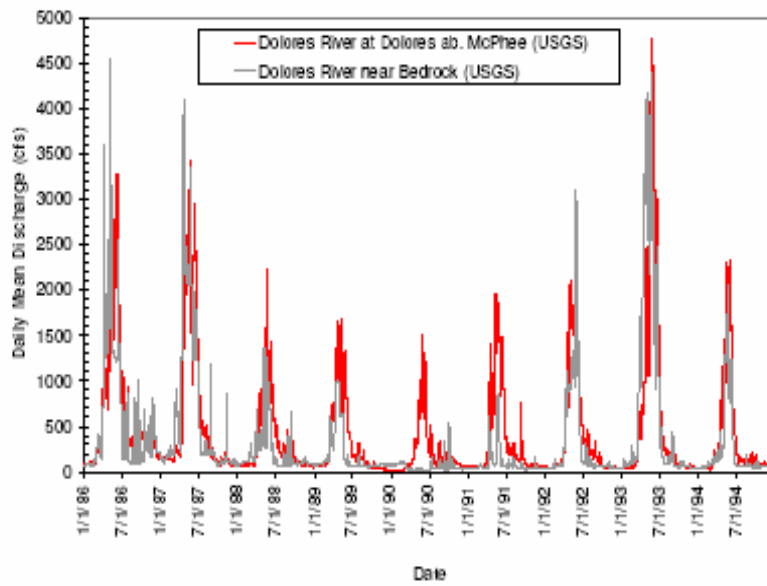


Figure 1 – Daily mean discharge from 1986 through 1994 at the Dolores gage above McPhee Dam and the Bedrock gage, 94 miles downstream of the dam.

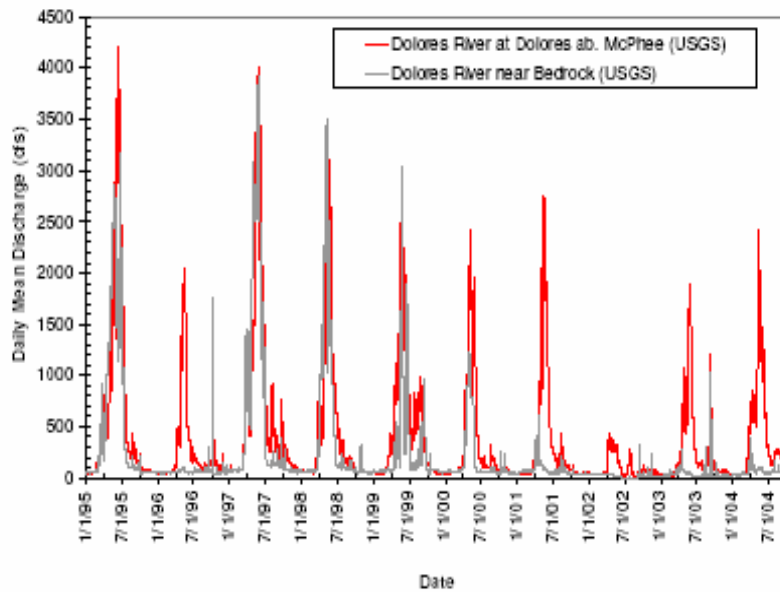


Figure 2 – Daily mean discharge from 1995 through 2004 at the Dolores gage above McPhee Dam and the Bedrock gage, 94 miles downstream of the dam.

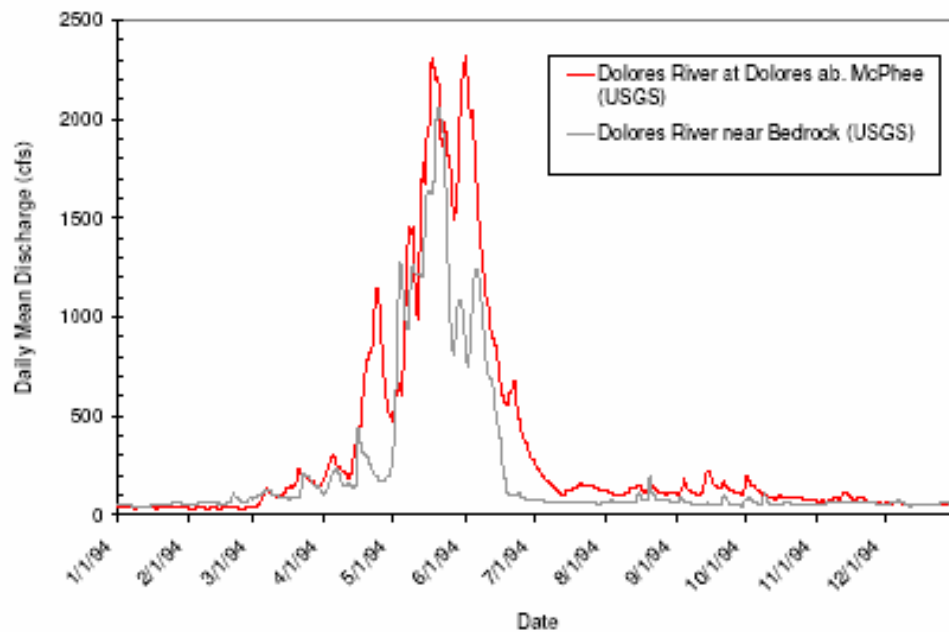


Figure 3 – Daily mean discharge for 1994 at the Dolores gage above McPhee Dam and the Bedrock gage, 94 miles downstream of the dam.

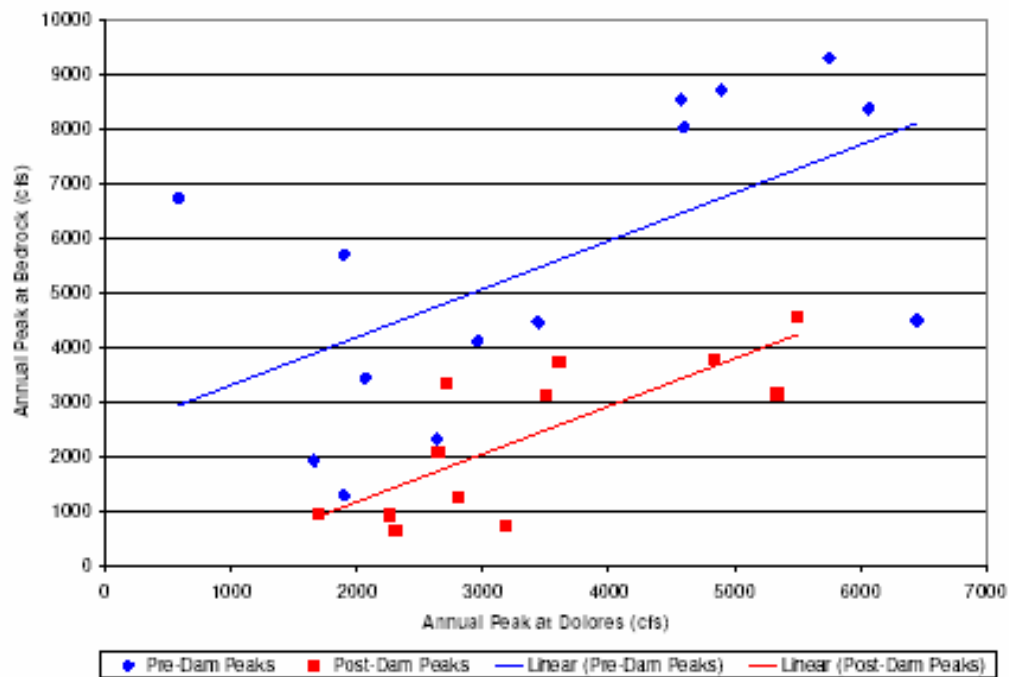


Figure 4 – Dolores (above McPhee Dam) vs Bedrock (below McPhee Dam) peakflows 1971-1984 (Pre-dam) and 1990-2001 (Post-dam) (created by David Graf, CDOW, 2005)

In order to evaluate the changes in the flow regime resulting from the construction of McPhee Dam, hydrologic data from the Bedrock gage and the gage near Cisco, UT on the Dolores River were analyzed using the Indicators of Hydrologic Alteration (IHA) software (Richter et al., 1996). The analysis was performed by David Graf at CDOW (2005). IHA evaluates changes in flow regime by comparing pre-impact mean daily flows with post-impact mean daily flows. Results of the analysis include 32 parameters that statistically characterize the flow regime and differences between the pre-impact and post-impact periods. The summary statistics can be used to understand changes in geomorphically important hydrologic processes along the Dolores River as a result of McPhee Dam. Of the 32 parameters summarized by the IHA software, the most geomorphically important are the magnitude and duration of annual maximum flows and the frequency and duration of high pulses (high flows that are less than the annual maxima). At the Bedrock gage, the parameters exhibiting the greatest impact by the dam are the annual maximum flows (36 to 41% decrease) and the duration of the high pulse (60% decrease) (Table 1). The Cisco gage is below the input of the San Miguel River, the largest tributary to the lower Dolores River. The impacts of the dam on the flows at the Cisco gage are not as significant as at the Bedrock gage (Table 2). The greatest change is in the one and three day maximum flows (13% decrease).

Table 1 – IHA Results for the Dolores River at Bedrock gage.
Pre-Impact Period: 1918-1983 (18 years)
Post-Impact Period: 1984-2003 (20 years)

	Mean Value for Pre- Impact Period	Mean Value for Post- Impact Period	Magnitude of difference between pre and post- impact period	% change between pre and post- impact periods
April - mean monthly flow (cfs)	1215	867	-348	-29
May - mean monthly flow (cfs)	2136	1289	-848	-40
June - mean monthly flow (cfs)	1444	670	-774	-54
July - mean monthly flow (cfs)	265	137	-127	-48
1-day maximum (cfs)	3811	2296	-1514	-40
3-day maximum (cfs)	3541	2180	-1361	-38
7-day maximum (cfs)	3151	2016	-1135	-36
30-day maximum (cfs)	2390	1464	-926	-39
90-day maximum (cfs)	1667	981	-686	-41
High pulse* duration (days)	18	7	-11	-60

*The high pulse level is 1508 cfs (mean of pre-period 95th percentile flow)

Table 2 – IHA Results for Dolores River at Cisco, UT gage.

	Pre-impact period - 1951 - 1983 (33 years) Post-impact period - 1984 - 2003 (20 years)			
	Mean Value for Pre- Impact Period	Mean Value for Post- Impact Period	Magnitude of difference between pre and post- impact period	% change between pre and post- impact periods
April - mean monthly flow (cfs)	1960	2011	51	3
May - mean monthly flow (cfs)	2971	3016	45	2
June - mean monthly flow (cfs)	2078	1705	-374	-18
July - mean monthly flow (cfs)	649	579	-70	-11
1-day maximum (cfs)	5730	4978	-752	-13
3-day maximum (cfs)	5433	4746	-686	-13
7-day maximum (cfs)	4886	4469	-417	-9
30-day maximum (cfs)	3574	3345	-229	-6
90-day maximum (cfs)	2451	2308	-143	-6
High pulse* duration (days)	15	20	5	31

*The high pulse level is 2320 cfs (mean of pre-period 95th percentile flow)

Channel-forming discharge

The channel-forming discharge or the flow that maintains the channel morphology and planform are typically estimated in various ways. One estimation method is to determine the flow that fills the available channel cross section, referred to as the bankfull discharge (Wolman and Leopold, 1957). Another method is to determine the flow that carries the most sediment, which is called the effective discharge (Wolman and Miller, 1960). Another way to estimate the channel-forming discharge is to determine what discharge is necessary to mobilize the sediment particles comprising the bed material of the channel. A correlation between both the bankfull and effective discharge and the 1-2 year recurrence interval flood has been found in many rivers, particularly those in humid, temperate areas (Wolman and Leopold, 1957, Wolman and Miller, 1960). The designation of the 1-2 year flow as the channel-forming discharge is an oversimplification in many respects, because channel form is product of a range of discharges (which may include the bankfull discharge) rather than a single formative discharge and of the temporal sequence of high-flow events (Knighton, 1998). In river segments where the channel boundary is controlled by bedrock or other highly resistant material, high-magnitude, low frequency floods (>>1-2 year) may represent the channel-forming discharge (Knighton, 1998). This caveat is particularly relevant to bedrock-controlled segments of canyon reaches of the Dolores River, where floodplains are largely absent and the canyon walls comprise the channel boundaries. In 1990, BLM performed an assessment of the pre- and post-dam flow regimes of the Dolores River and the impacts of changes in the flow regime on channel morphology (USDI BLM, 1990).

Bankfull discharge estimates for the Dolores River from McPhee Dam to Bedrock ranged from 2,000 cfs at Bradfield Bridge (11 miles below McPhee Dam) to 2,500 cfs at Bedrock (94 miles below McPhee Dam). The 1.5-year recurrence interval peak flow increased from 2,589 cfs at the Dolores gage (which is located upstream of McPhee Dam and irrigation diversions) to more than 3,000 cfs at the Bedrock gage (Table 3).

Table 3 – Bankfull Flow Estimates at USGS Gaging Sites (from USDI BLM 1990)

Station	Period of Record	1.5-year Recurrence Interval Peak Flow (cfs)	1.5-year Recurrence Interval 7-day High Flow (cfs)
At Dolores	1895-1903, 1910-12, 1921-82	2,589	2,045
Near McPhee	1939-1952	2,964	2,110
At Bedrock	1918-22, 1972-83	3,058	1,808

More recent analyses of the streamflow record at the Dolores gage above McPhee Dam reveals a lower estimate of the 1.5-year discharge. The 1.5-year discharge at the Dolores gage based on 92 years of record is 2,200 cfs (Figure 5). Since construction of McPhee Dam, the magnitude of the 1.5-year flow directly downstream of the dam has been reduced to less than 1,000 cfs (Figure 6), less than half the 1.5-year flow upstream of the dam.

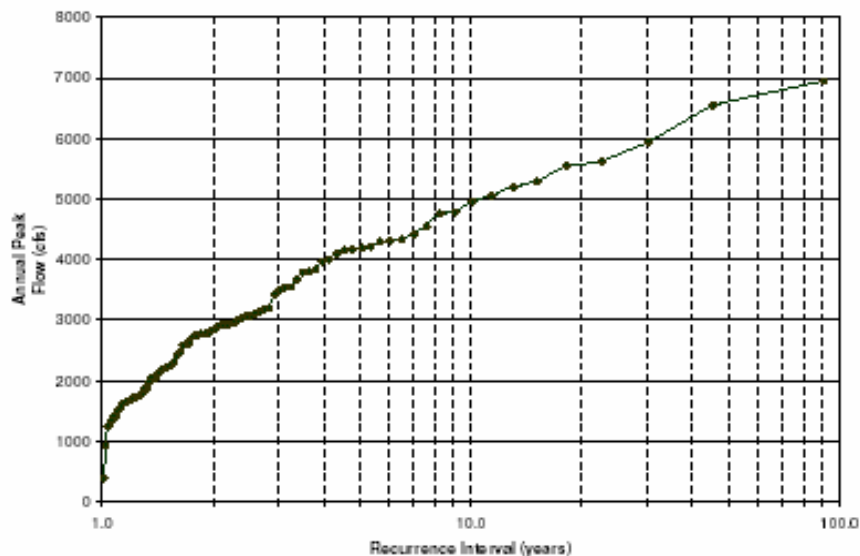


Figure 5 - Annual Peak Frequency Plot - Dolores River at Dolores (92-year record) (Developed by David Graf, CDOW, 2005)

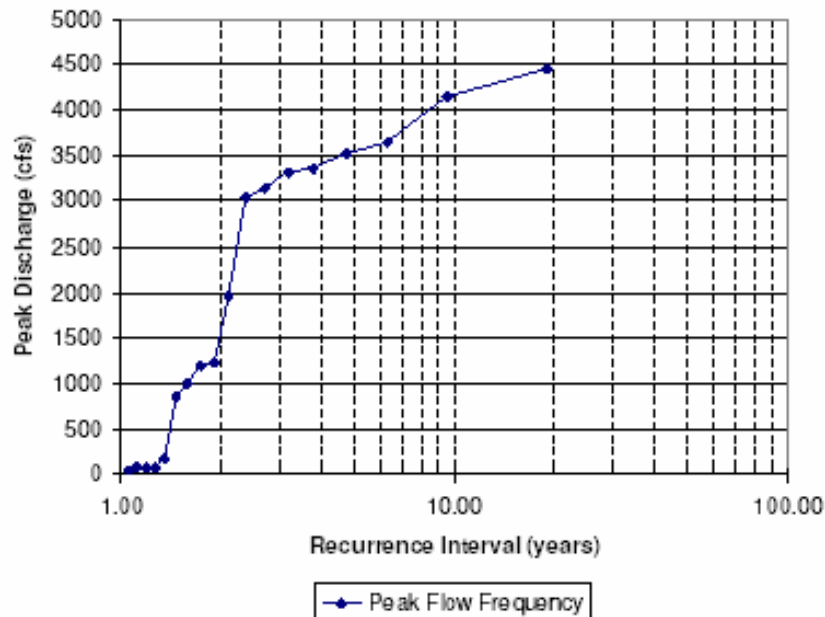


Figure 6 - Peak Flow Frequency below McPhee Reservoir (1986 – 2003) – Colorado DWR gage. (Developed by David Graf, CDOW, 2005)

BLM (1990) also estimated the flows necessary to mobilize the bed material between Bradfield Bridge and Bedrock. This report suggested that a flow of 2,000 cfs is required to move the median bed material size (D50), and that a flow of 7,000 cfs is necessary to move most of the bed materials (USDI BLM, 1990). These results suggest that prior to construction of McPhee Dam, the median size bed material moved on the average every 1 to 2 years, and the larger bed material sizes moved about once every 5 to 10 years (USDI BLM, 1990). Recent estimates of bankfull discharge have been developed for the Dolores River in Big Gypsum Valley (Figure 7) (Richard and Anderson, Preliminary Results) using HEC-RAS river modeling software to determine at the discharge associated with floodplain inundation. Twenty five cross sections were established throughout a 1.9-mile reach (Figure 8). Preliminary results suggest that flows of at least 2,000 cfs are necessary to inundate the floodplains and that 1,000 cfs does not flow onto the floodplain surfaces. Further calibration and validation of the model are necessary to further refine the bankfull discharge estimates. The results correlate well with BLM's (1990) estimate of 2,200 cfs bankfull discharge below Disappointment Creek. Cross sections BG19 and BG20 (BG = Big Gypsum, see Figure 9 and Figure 10) provide an example of the modeled floodplain inundation. Flows of 1,000, 2,000 and 3,000 cfs were modeled in HEC-RAS as well as a calibration flow of 60 cfs (Figure 11 and Figure 12). The model results were combined with a digital elevation model of the channel and adjacent topography to show areas inundated at different flow rates (Figure 13, Figure 14 and Figure 15). The resulting inundated areas show that 1,000 cfs does not inundate the floodplain. Similar results were obtained in other areas of the Big Gypsum reach, for example at cross sections BG16 and BG17 (Figure 16 and Figure 17).



Figure 2= Location of Big Gypsum Reach on the DeJoria River, Colorado.

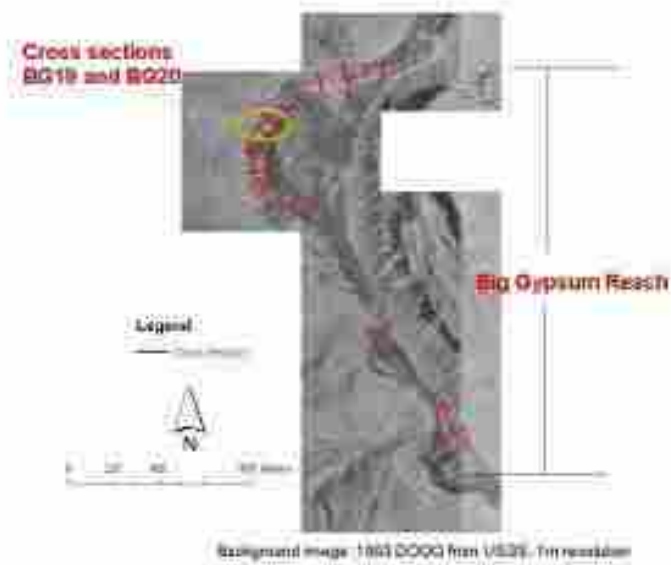


Figure 8- Big Gypsum reach and location of cross sections BG19 and BG20.



Figure 9- Floodplain in vicinity of cross sections 19 and 20. View is looking upstream from right bank of river.



Figure 10 – Approximate location of cross sections 18 through 20 and the adjacent floodplain area. View is looking downstream from a bluff on the left bank of the river.

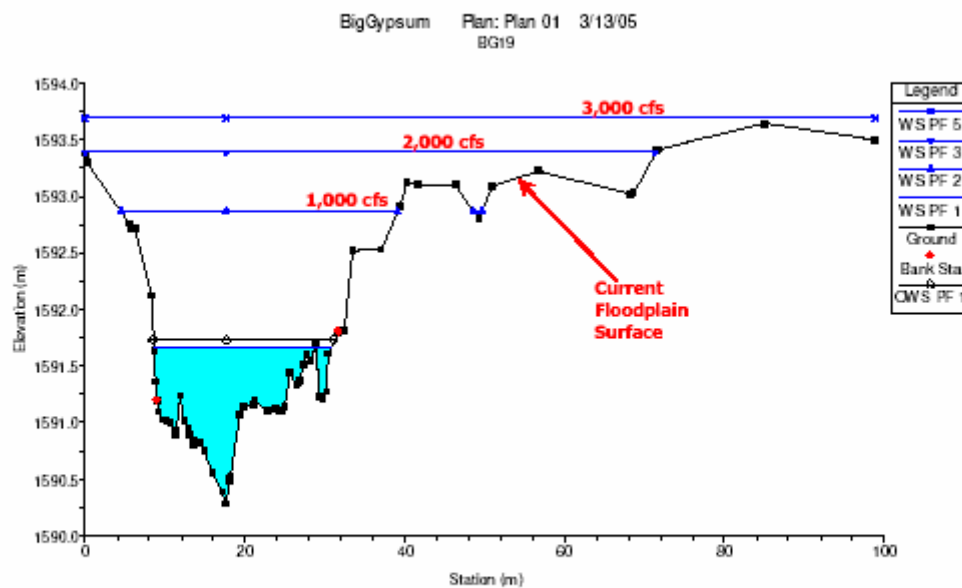


Figure 11 – River Station 756 (measured in meters from downstream end of reach), Cross Section BG19. (BG = Big Gypsum Reach, WS PF 5 = 3,000 cfs, WS PF 3 = 2,000 cfs, WS PF 2 = 1,000 cfs, WS PF 1 = 60 cfs, OWS PF1 = observed water surface for 60 cfs calibration flow)

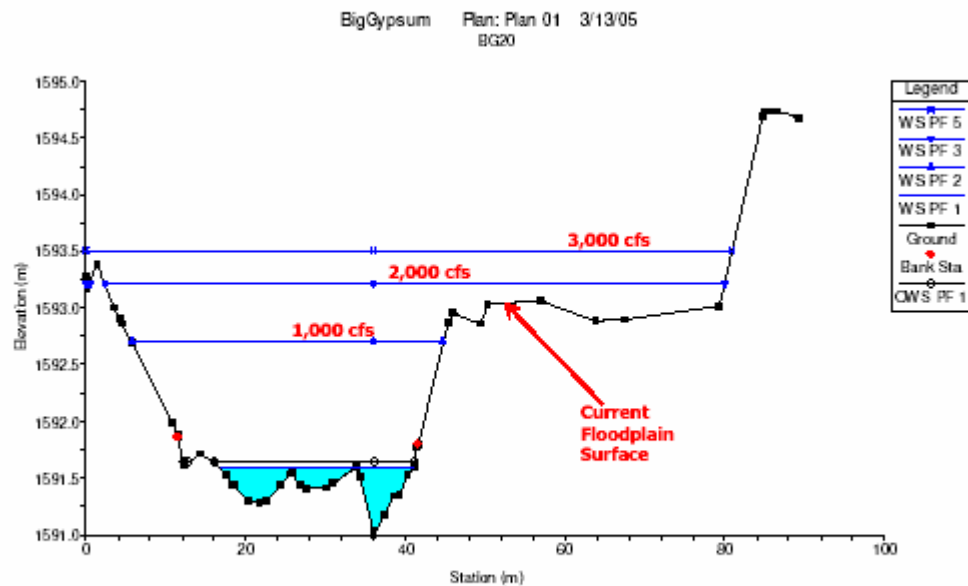


Figure 12 – River Station 683 (measured in meters from downstream end of reach), Cross Section BG20. (BG = Big Gypsum Reach, WS PF 5 = 3,000 cfs, WS PF 3 = 2,000 cfs, WS PF 2 = 1,000 cfs, WS PF 1 = 60 cfs, OWS PF 1 = observed water surface for 60 cfs calibration flow)

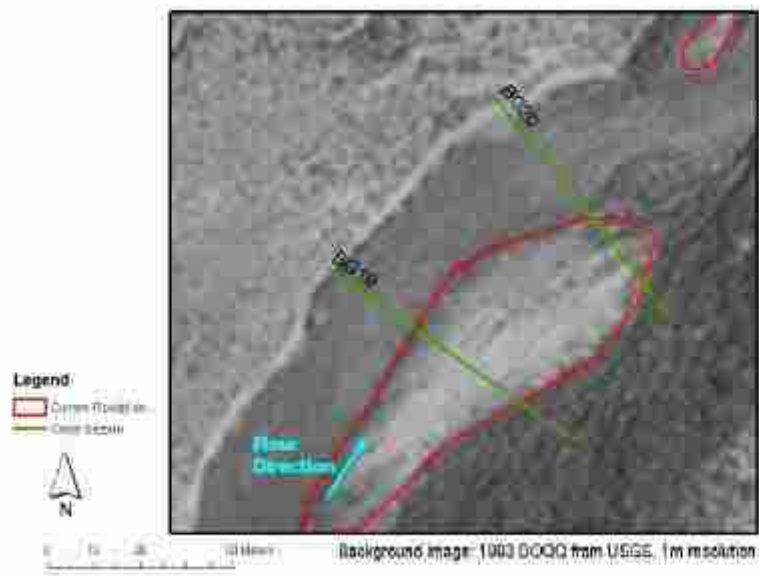


Figure 13 – Location of cross sections BG19 and BG20 and adjacent floodplain surface.

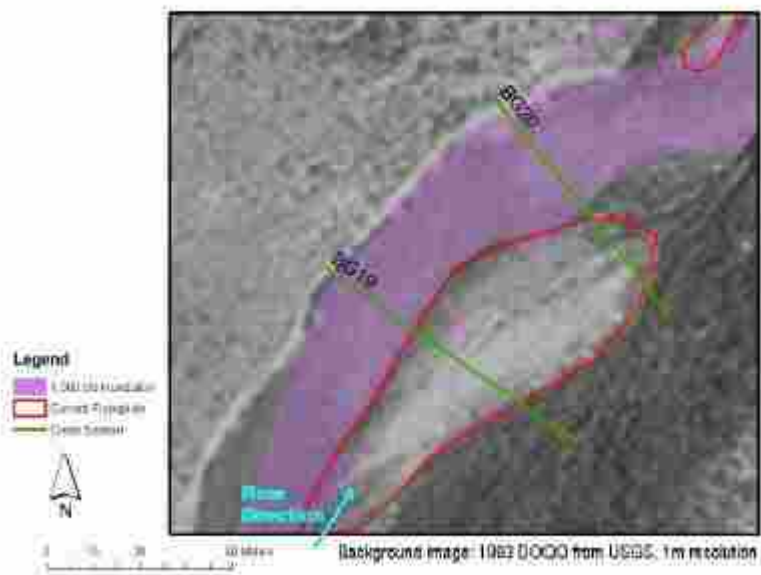


Figure 14 - foundation area for a modeled flow of 1,000 cfs at cross sections BG 19 and BG 20.

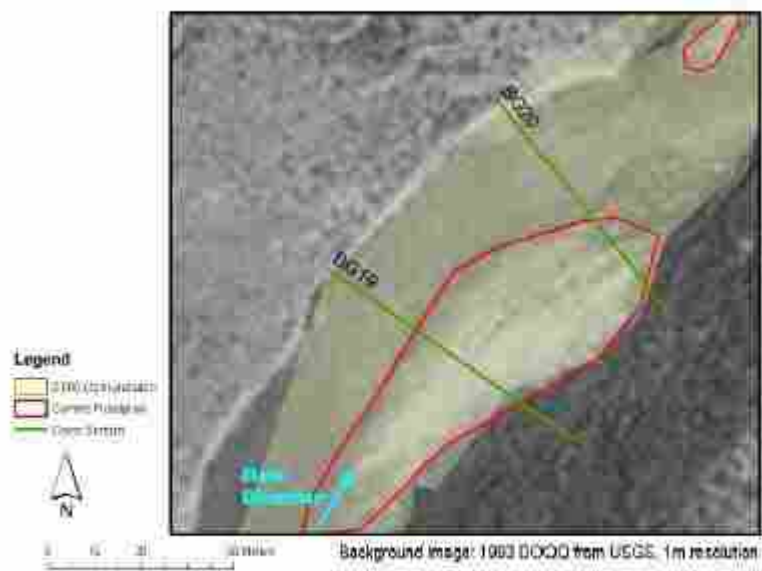


Figure 15 - foundation area for a modeled flow of 2,000 cfs at cross sections BG 19 and BG 20.

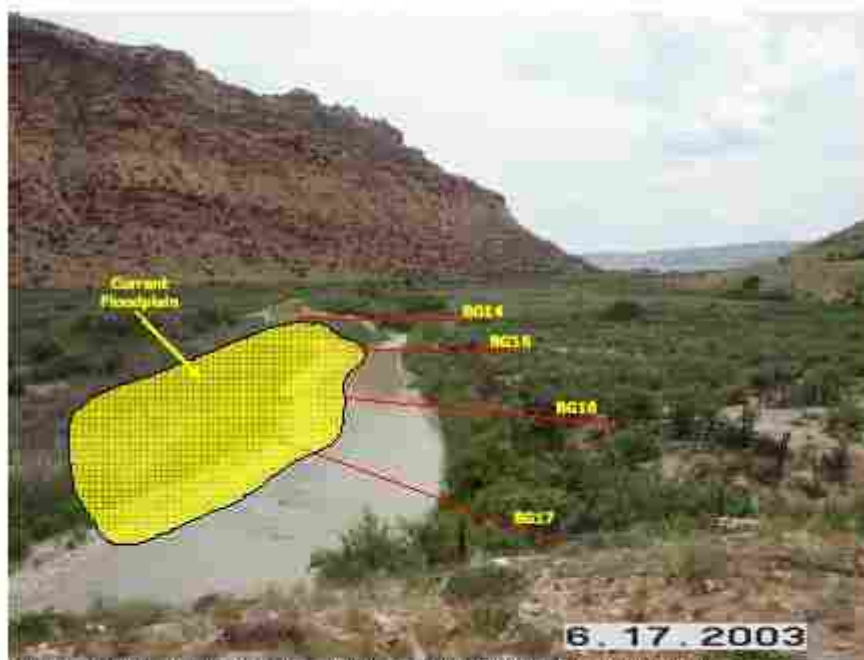


Figure 16 – Current floodplain at Cross sections 14 and 15. View is upstream from bluff on left bank of river.



Figure 17 – Surveying the floodplain downstream of cross section 15. Note vegetation establishment. View is looking downstream on right bank of river. June 17, 2003. Discharge at Redrock weir = 10 cfs.

walls, producing changes in bed sediment characteristics. These potential changes are discussed below in the section on sediment dynamics. Reductions in sediment transport capacity also may have reduced the frequency of pool scour and the reworking of gravel deposits on riffles, resulting in an overall simplification of bed morphology. Further, the reduced frequency with which the Dolores River attains discharges sufficient to overtop its banks suggests that channel floodplain connectivity and flow-driven adjustment of floodplain processes has declined. The magnitude of these changes has likely varied in reaches downstream of McPhee Dam, depending on the background geomorphic conditions of the reaches and on the availability of unregulated high flow inputs from tributary basins. Because tributary flow inputs are limited between McPhee Dam and the San Miguel River confluence, the effects of reduced high flows have likely been concentrated upstream of the San Miguel. The San Miguel River is a large tributary with minimal flow regulation and continues to deliver high flows to the lower Dolores River. The San Miguel River therefore likely dampens the effects of McPhee Dam and limits the magnitude of changes in geomorphically important flows downstream of its confluence with the Dolores, as illustrated by the IHA analysis discussed above.

Geomorphic changes associated with flow modification therefore have likely been smaller downstream of the San Miguel than upstream. Factors including bed gradient, confinement, and tributary influences mediate the sensitivity of reaches between McPhee Dam and the San Miguel to flow-induced geomorphic changes. Overall, lower-gradient, unconfined reaches are those that are most susceptible to geomorphic changes such as fining of bed material, channel narrowing, and channel simplification. In the Dolores River, such reaches include the reach from McPhee Dam to the upstream end of Dolores Canyon (3-4 miles downstream of Bradfield Bridge) and portions of the river flowing through Big Gypsum and Paradox Valleys. In addition, reaches downstream of sediment-producing tributaries, especially Disappointment Creek, have likely experienced flow induced geomorphic changes due to reductions in the mainstem river's ability to rework and transport tributary sediment inputs. In contrast, high-gradient, confined, and bedrock-controlled canyon reaches may be relatively less susceptible to geomorphic changes. It is likely that even in these reaches, channel narrowing due to growth of bars and vegetation encroachment, as well as fining of bed sediment, has occurred.

Sediment dynamics in the Dolores River basin

Sediment dynamics, including the timing and magnitude of sediment supply to streams, sediment transport within streams, and the size, heterogeneity, and stability of substrates, have an important influence on channel form and are an important driver of physical habitat structure (Waters 1995). Sediment dynamics are especially important to bottom-dwelling aquatic organisms, which are often associated with specific sediment environments (Allan 1995, Palmer et al. 2000). Alteration of sediment regimes can degrade habitat through burial or scour, changes in substrate heterogeneity and stability, and altered channel form. In addition, interactions between alterations in flow and sediment regimes have important effects on aquatic biota and implications for the conservation of many at-risk aquatic species (Osmundson et al. 2002, Hart and Finelli 1999). The geology, climate, soils, and vegetation characteristics of the Dolores River basin result in a substantial increase in sediment yield, in both absolute and per acre terms, from the upper to the lower basin. Data from gaging stations indicates that total suspended sediment

(TSS) increases from an average of 182,000 tons/yr (0.34 tons/acre/yr) above McPhee Dam (USDI BLM 1990) to 2,524,000 tons/yr (0.86 tons/acre/yr) at the Dolores River at Cisco (USGS gage 0918000; 1949-1964).

Estimated sediment yields in the watershed upstream of McPhee dam are very low (<0.2 ac-ft/mi²/yr in most of the drainage area), which is in order of magnitude lower than downstream portions of the Dolores River basin) (USDA, 1972). Areas downstream of McPhee Dam have a more arid climate, sparser vegetation, and less resistant rock types than upstream areas, resulting in increased erosion potential. Much of the Dolores River downstream of McPhee Dam is flanked by rock outcrops (mostly sandstone, with some shale outcrops) and/or areas with very shallow soils. Rockfall from canyon walls is likely an important source of coarse sediment to the river, and the estimated sediment yield from these areas ranges from <.2-1 acft/ mi²/yr (USDA, 1972). Substantially higher sediment yields (1-3 ac-ft/mi²/yr) are found in the three large structural valleys that separate the canyon reaches of the Dolores River: Disappointment Valley, Big Gypsum Valley, and Paradox Valley (USDA, 1972). Among these, Disappointment Valley delivers the largest amount of sediment to the mainstem Dolores River due to the perennial flow in Disappointment Creek, which enters the mainstem at River mile 124 near Slickrock (USDI BLM, 1990). Much of Disappointment Creek's 350-mi² drainage area is underlain by Mancos Shale, which is mantled by shallow, easily eroded soils that produce considerable sheet and gully erosion (USDA, 1972).

The above description of sediment yield characteristics in different portions of the Dolores River basin provides a basis for analyzing the likely effects of McPhee Dam on sediment dynamics in the Dolores River. Dams can potentially alter downstream sediment dynamics by blocking sediment supply to downstream reaches and/or by reducing the frequency and magnitude of sediment-mobilizing flows (Ligon et al., 1995). In terms of the first of these potential impacts, available evidence suggests that the effect of McPhee Dam on downstream sediment supply to the Dolores River is small. As discussed above, the geology and climate of the watershed upstream of McPhee Dam result in a low rate of natural sediment delivery to channels. In addition, field observations of McPhee Reservoir suggest that minimal sediment accumulation has occurred in the reservoir since dam closure in 1984, although no formal measurements of reservoir sedimentation have been completed. Moreover, natural and anthropogenic sediment delivery to the Dolores River downstream of McPhee Dam, including from Disappointment Creek and other tributaries, is substantial, likely overcompensating for any potential reduction in upstream sediment supply. McPhee Dam has had substantial effects on the frequency and magnitude of sediment mobilizing flows, however, likely resulting in changes in bed sediment composition and in channel form. Data on bed-substrate sizes in the Dolores River are limited, but geomorphic principles and analysis of flow modification suggests that flow modification has likely produced an overall fining (reduction) of bed sediment sizes and a decreased frequency of sediment mobilization. Historically, sediment entering the main stem Dolores River from tributary basins

and/or from canyon walls would be mobilized and reworked with a frequency proportional to the size of the sediment. For example, fine sediments delivered from tributaries to the mainstem would likely be transported downstream at low to moderate discharges under unregulated conditions, but such sediments now are deposited on the channel bed and result in siltation. Such siltation increases the embeddedness of bed materials, as fine sediments infiltrate into the

interstices of coarser bed materials such as gravels and cobbles. Coarser sediments (e.g., gravel-size sediment) deposited in the mainstem from side canyons form tributary bars, and most of the rapids in the Lower Dolores River are produced by such depositional features (USDI BLM, 1990). Whereas at least some size fraction of these bar deposits would likely be mobilized on an annual basis by high flows under unregulated conditions, these coarse materials are likely rarely transported under current conditions, potentially causing steepening of rapids. This potential change in the stability of tributary bar deposits is analogous to the conditions in the Colorado River in Grand Canyon associated with reductions in peak flows by Glen Canyon Dam. The effects of flow modification-induced changes in downstream sediment characteristics may have been exacerbated by increases in sediment supply above natural levels due to agriculture and other land uses. A 1972 report suggested that overgrazing had depleted vegetation cover and increased soil erosion in many arid rangeland areas (USDA, 1972). Because of the dry climate and slow rate of recovery of vegetative cover in this area, grazing induced increases in erosion likely persist in the lower Dolores River basin. Bank erosion associated with sport fishing and rafting may locally increase sediment inputs. Roads and other land uses have also likely increased sediment inputs to river channels, but these sources are likely minor compared to natural sediment yields. Overall, grazing, recreation, roads, and other land-use activities have likely had a small effect on geomorphic processes compared to the effects of flow modification.

Key data gaps

As mentioned above, much of the information provided here about geomorphic conditions in the Dolores River and about potential human effects on those conditions is based on previous reports and other remote methods. Data are needed on bed substrates sizes and cross-section geometry (especially repeat surveys of previously measured cross-sections) in order to develop scientifically based estimates of the discharge magnitudes required to mobilize sediment of various size fractions and perform other geomorphic work. Additional historical analysis of repeat sequences of aerial photographs in relation to flow data would elucidate relationships between geomorphic processes and high-flow magnitude, frequency, and duration, although available aerial photographs may be of insufficient resolution for such analysis. Data on geomorphic conditions in canyon reaches is especially sparse due to the inaccessibility of many of these reaches.

Summary

This analysis suggests that reductions in high flows below McPhee Dam are a primary cause of geomorphic changes in the Dolores River downstream of the dam. The primary changes that have occurred due to flow modification have likely included:

- changes in channel dimensions, including narrowing and reduced depth;
- growth of lateral and mid-channel bars associated with reduced sediment
- mobility and encroachment of riparian vegetation;
- changes in the size, embeddedness, and mobility of bed sediments;
- simplification of bed morphology; and
- reduced channel-floodplain connectivity.

The magnitude of these changes has likely varied in downstream reaches, depending on the background geomorphic conditions of the reaches and on the availability of unregulated high flow inputs from tributary basins. Because tributary flow inputs are limited between McPhee Dam and the San Miguel River confluence, the effects of reduced high flows have likely been concentrated upstream of the San Miguel. High flow inputs from the San Miguel River have limited the magnitude of changes in geomorphically important flows in the lowermost reaches of the Dolores River and have therefore likely limited the geomorphic changes associated with flow modification.

Factors including bed gradient, confinement, and tributary influences mediate the sensitivity of reaches between McPhee Dam and the San Miguel to flow-induced geomorphic changes. Overall, the reaches that are most susceptible to geomorphic changes in the Dolores River are those with shallow gradients and wide valley bottoms, including the reach from McPhee Dam to the upstream end of Dolores Canyon (3-4 miles downstream of Bradfield Bridge) and portions of the river flowing through Big Gypsum and Paradox Valleys. In addition, reaches downstream of sediment-producing tributaries, especially Disappointment Creek, have likely experienced flow-induced geomorphic changes due to reductions in the mainstem river's ability to rework and transport tributary sediment inputs.

Other anthropogenic factors influencing geomorphic processes in the Dolores River have historically included roads, grazing, and recreation. Valley-bottom roads constrain channel migration along certain reaches, and grazing-induced reductions in vegetative cover have likely increased sediment inputs to the lower Dolores River. Bank erosion associated with sport fishing and rafting may locally increase sediment inputs. Overall, however, these activities have likely had a small effect on geomorphic processes compared to reduction in the magnitude, frequency, and duration of high flows caused by McPhee Dam.

The preceding analysis is intended to serve as the basis for future discussion of approaches to improving ecological conditions in the Dolores River downstream of McPhee Dam. This analysis suggests that management of high flows, subject to legal, operational and safety constraints, has the greatest potential for maintaining or restoring geomorphic processes and the aquatic habitats shaped by these processes. Geomorphologists from the Core Science Team will work with the Dolores River Dialogue to determine the magnitude, frequency, and duration of flows needed to maintain pool scour, rework coarse sediment, flush fine sediments, limit vegetation encroachment and growth of bars, and perform other ecologically important geomorphic work.

Wilcox and Richard

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Dolores River Dialogue Riparian Vegetation Analysis

David Merritt, June 3, 2005

The purpose of the following discussion is to give the reader a general familiarity with some basic principles in riparian plant ecology, to relate these principles to the riparian vegetation of the Dolores River, and to provide recommendations for modifying streamflow and fluvial processes to achieve specific management goals. The first section, River processes, riparian vegetation and flow components, provides concepts that relate the life-history of riparian plants to attributes of the hydrologic regime (flow components). The second section, Riparian vegetation of the Dolores River, highlights the few studies that have taken place along the Dolores River that have described the riparian plant communities and changes in them over the past couple of decades. The third section, An approach to inventorying the riparian vegetation of the Dolores River, discusses some considerations for developing an inventory and monitoring system for the Dolores River so that current condition, trends through time, and responses of riparian plants to management activities can be quantified. The last section, Changes in flow regime on the Dolores River, flow components, and vegetation, discusses the biologically relevant changes in flow regime that have occurred since the operation of McPhee dam began in 1984 and provides some discussion of the changes in riparian plant community composition and population structure that are likely to have occurred along the Dolores River as a result of changes in physical processes. Flow recommendations are provided.

River processes, riparian vegetation and flow components

Riparian vegetation is responsible for many of the human-valued ecological services provided by rivers. Plants growing along rivers provide habitat structure, shelter and roosting areas for insects and animals, and inputs of carbon (leaves and wood) and nutrients that support aquatic communities (insects, amphibians, and fish). Plants also contribute to water quality through filtering and uptake of nutrients, chemicals, and metals, and filtering of sediment through stabilizing stream banks and channel features. Plant communities growing along rivers are disproportionately species rich compared to upland plant communities (Naiman et al. 1993). Maintaining or restoring the processes that support native plant communities and disfavor exotic species is becoming increasingly important in river management worldwide.

The availability of water along rivers is one factor that contributes to the lush, productive and diverse vegetation that characterizes river valleys in the western US relative to surrounding uplands. The most extensive deciduous forests in arid regions of the West occur along rivers. The formation and maintenance of riparian (streamside) forests and native-dominated woodlands require specific processes that facilitate their establishment and development: occasional overbank flooding, the redistribution of sediment on bars and across the floodplain, and hydrograph attributes that favor seedling recruitment. In the absence of such processes, Western riparian communities shift from those dominated by disturbance adapted species such as cottonwood, willow, and annuals to later successional species such as perennials and woody species (eg., box elder, rabbitbrush, wild privet, big sage, western juniper).

A variety of other factors influence composition and structure of the vegetation that grows in riparian areas. An understanding of the factors that lead to the range of possible vegetation types along a particular reach of river can be helpful in evaluating relative trade offs -- the cost of

modifying driving factors such as streamflow and the potential benefits of achieving specific goals with respect to the function and form of riparian areas.

A variety of factors influence the structure and composition of riparian vegetation growing along a particular reach of river. Geologic setting, short-term weather patterns and long-term climatic fluctuations, and direct human activities all influence the composition and dynamics of riparian plant communities. For example, the width of a river valley between confining canyon walls and depth of valley fill influence groundwater patterns and determine the extent of available habitat for riparian plants, natural wet and dry cycles can be reflected for centuries in the population structure of woody vegetation, and the building and maintenance of roads and other human-activities can serve to remove, constrain, or cause significant shifts in riparian vegetation.

Riparian plants are limited by many of the same factors that influence plants growing in uplands: fire, grazing and insect herbivory, water availability-drought, competition with other plants, and availability of suitable habitat. Plants growing along rivers are also subjected to disturbances that are unique to streamside areas, primarily fluvial processes (those related to flowing water such as scour and burial) and stresses associated with flooding (anaerobic conditions and periodic submergence). As a consequence, riparian vegetation along free-flowing arid-land streams is often dominated by species adapted to (and in some cases reliant upon) such processes. Because fluvial processes distinguish riverine habitats from other ecosystems, a discussion of the processes that structure and support riparian vegetation hinges upon an understanding of the interplay between hydrology, geomorphic processes, and the life history attributes (requirements at various stages of development) of plants.

For plants, each stage of development from seed to reproductively active adult must be met (at least occasionally) for self-perpetuating populations to persist. Completion of the life cycle for plants requires that: 1) suitable habitat must occasionally become available (newly formed or cleared of existing vegetation) for colonization, 2) seeds must reach such habitat and germinate, and 3) seedlings must persist and mature to reproductive age (Figure 1). The same principle applies for species that reproduce vegetatively (by root sprouts, stolons, etc.). For many species that are only found in riparian areas, these stages are tied to some aspect of river flow regime. In the arid West few obligate riparian species can persist solely on the water provided by the local climate; groundwater and or streamflow is necessary at certain times during the year. Of course, extreme or prolonged drought affects both riparian and upland species, whereas longterm human-caused dewatering of streams may result in riparian plant communities becoming more similar over time to the plant communities characteristic of adjacent uplands.

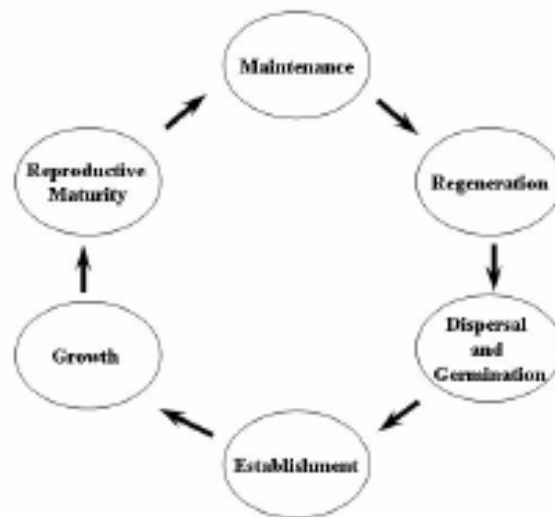


Figure 1. Stages in the life history of plants. Stages are tied to some component of the hydrologic regime for many riparian species. Interruption of any stage in life-history can influence the population structure of the species.

Flow regime, the seasonal distribution of water along rivers, can be simplified into a small number of “flow components” that can be directly linked to life stages of plants (Figure 2). The magnitude and duration of floods (high flow), the duration and severity of drought (low flow), the rate of change in flows, and the timing of flows relative to the life stages of riparian plants are the four main components of flow that will be discussed.

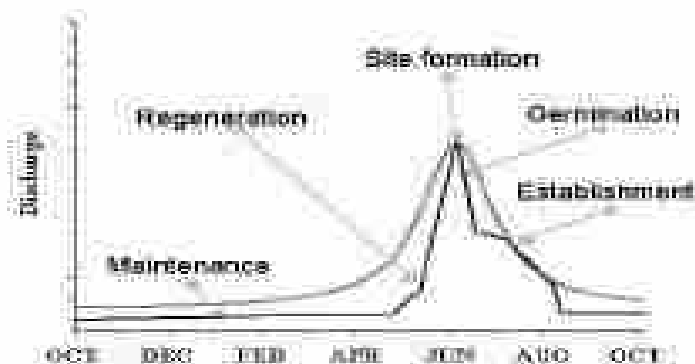


Figure 2. Stages in the life-history of a particular species may be linked to hydrograph attributes (flow components). Dark hydrograph line illustrates a strategically designed hydrograph developed to accommodate a particular species while maximizing efficiency of water use. Structured hydrographs could be used occasionally (only when sufficient volumes of water are available) to help achieve some defined management goal.

High Flows

Fluvial disturbance associated with high flows serves to maintain heterogeneity in riparian habitats by influencing the distribution and characteristics of sediment deposits in the channel and across the floodplain. The heterogeneous habitat maintained by fluvial processes contributes to the high species richness in riparian areas (Naiman et al. 1993, Tiegls et al. 2005). Because large (i.e., overbank) floods do most of the geomorphic work along rivers, many of the characteristics of the physical habitat --such as the shape of the channel and the particle size of the substrate forming the bed and banks of the stream-- are structured to some degree by flooding. One large flood can create habitat that is maintained by smaller floods and serves as sites of plant colonization for decades (Baker 1988, Friedman and Lee 2002, Cooper et al. 2003, Lytle and Merritt 2004). The texture of sediment deposited by floods influences water availability to plants as it governs the water holding capacity of the substrate. Flooding also results in the removal or burial of plants or portions of plants and results in open regeneration sites for pioneer species to colonize. High flows may also prevent encroachment of vegetation into the channel (Friedman and Auble 1999). Flooding also recharges floodplain soils and shallow alluvial aquifers, facilitates decomposition of leaf litter and the availability of nutrients on the floodplain, and increases the productivity of riparian vegetation (Merritt and Cooper 2000, Ellis et al. 1998).

Cottonwood (*Populus deltoides* and *P. angustifolia*) are valued riparian forest species. Cottonwood colonize recently disturbed open patches of moist sediment. The processes that form the types of sites that cottonwood can become established upon include those associated with flooding, such as overbank sediment deposits and point bar development associated with lateral channel migration (Bradley and Smith 1984, Scott et al. 1996). Channel narrowing can also facilitate or be facilitated by cottonwood establishment (Friedman et al. 1996, Scott et al. 1996) or establishment of other species such as tamarisk (Graf 1978, Allred and Schmidt 1999).

Reducing the frequency of high flows can cause shifts in species composition in riparian areas by favoring those species that are more aggressive competitors (i.e., some non-native plant species) and causing a reduction in those species adapted or otherwise able to colonize following such disturbances. One prominent example is the widespread decline of cottonwood forests throughout western North America in the 20th century (Rood and Mahoney 1990). Reduction in fluvial processes (i.e., overbank flooding, lateral channel migration) associated with reductions in peak flows due to water storage has been implicated in reduced recruitment and reductions in extent of cottonwood forests (Bradley and Smith 1986, Rood and Mahoney 1990, Cooper et al. 1999, Braatne et al. 1996). There are cases of cottonwood forests temporarily expanding in extent in response to river damming in areas where shallow groundwater facilitated colonization and channel narrowing (Johnson 1994).

Often, reductions in high flow initiate a directional vegetation response: shifts in zonation of vegetation toward the channel. Reductions in the frequency of flooding can also result in other indirect changes such as an accumulation of salts in floodplain soils. Over long periods of time, the lack of occasional flooding can result in significant shifts in species composition of riparian vegetation due to changes in biological, physical, and chemical processes on the floodplain.

High flows are an important flow component for riparian vegetation in that they structure physical habitat, create open sites, facilitate recruitment, maintain active channels free of

vegetation, recharge and rinse alluvium forming the banks and floodplain, and they enhance productivity and fitness of existing vegetation.

Low flows

Low flows may be important in certain settings. For example, plants may be reliant upon shallow water tables (that are connected to streamflow) for maintenance and growth. Although some plants are able to deal with lowered water tables by utilizing moisture in the soil, plants obtaining most of their water from water table or capillary fringe (phreatophytes) may become water stressed and suffer from lowered water tables associated with stream dewatering. If atmospheric demands for water are higher than water available to roots, plants may respond by closing stomates (pores for gas exchange on leaves) and reducing transpiration. If the water deficit persists plants may become water stressed and sacrifice leaves or branches (Tyree et al. 1994). Intense or prolonged dewatering of streams can lead to desiccation and death (Rood et al. 2003). If water tables slope toward the stream, reduced low flows may steepen the water table surface and increase the depth to water across the floodplain, but dewatering a gaining stream is generally less detrimental to plants than dewatering a losing stream (Kondolf et al. 1987). One common long-term responses of vegetation to reduced water levels in a stream is for the zones of vegetation parallel to the stream shift downward and towards the formerly active channel (Auble et al. 1994, Nilsson and Svedmark 2002).

Individual species' tolerance for low moisture can determine dominance of certain species under conditions of low flow. Species with the capacity to withstand periods of drought or low flow may become dominant during drought or if water limitation persists. An example in Western riparian systems is the abundance of tamarisk relative to cottonwood in systems in which water is limiting or flood magnitudes have been significantly modified. Not only is tamarisk more drought tolerant than many other riparian species, but tamarisk is also tolerant of saline soils, which can limit native riparian species such as cottonwood. High salt levels in the soil exacerbate water stress of species not specifically adapted to salinity. Whereas tamarisk can germinate and grow in soils with extremely high salt levels, many native colonizers of riparian areas (i.e., cottonwood and willow) are inhibited by high salt concentrations (Shafroth et al. 1995). Seedlings of native species (cottonwood and willow) are most vulnerable to high concentrations of salt, but adults may persist and spread by vegetative means (such as by rhizomes and root sprouting). High salinity in valley bottoms and along streams is often associated with evaporation from shallow water tables. As was previously mentioned, in systems that experience occasional overbank flooding, lower salt levels may be maintained due to periodic flushing of floodplain substrate.

Increasing the stage and frequency of low flows relative to natural conditions can lead to encroachment and persistence of water-loving (hydrophytic species) near the channel or those species that can spread by roots, stolons, rhizomes or runners. Lush, marsh-like communities that form in areas with persistent high water tables may provide productive, high quality habitat, nutritious forage, and perform other beneficial functions (such as nutrient uptake; Stevens et al. 1995, Merritt and Cooper 2000). Elevated low flows can lead to higher rates of evapotranspiration of riparian vegetation and more water loss directly to evaporation from the

water surface, as the timing of low flow often coincides with hotter drier seasons in the West. In a general sense, increased low flows provide a more reliable source of water for plants during the late summer and autumn, which is beneficial to some species. In conjunction with reduced high flows, increased baseflows may lead to vegetation encroachment, channel narrowing, and reduced channel capacity.

Timing and rate of change in flow

Because some riparian species are adapted to the timing of certain flow components, altering the timing in flows from historic norms may be detrimental to those species (Lytle and Poff 2004). The historic timing of the peak flow on many streams in the West occurs in synchrony with the timing of seed release of some riparian species (particularly the Salices: cottonwood and willow) as well as the timing of native fish reproduction. Since cottonwood trees release seed over a short period of time in the spring, the seeds are dispersed at a time when newly created moist patches are just being exposed in conjunction with receding water levels.

Delayed timing in the peak may decouple the timing of seed release from the availability of suitable habitat. A delay in the peak by a few days or a week from the average date of the historic peak may not cause significant reductions in cottonwood establishment. But as the date of the peak deviates further and further from the historic peak, the likelihood of successful cottonwood seedling establishment diminishes. Furthermore, a delayed peak may foster establishment of later dispersing species such as tamarisk, which disperse seeds from mid summer through autumn in some regions in the Colorado Plateau (Cooper et al. 1999).

After cottonwood seeds germinate, seedlings require ample moisture to survive. If the material they are rooted into is coarse textured and well-drained, the water table must recede at a rate that does not exceed that of the rate of root growth of cottonwood seedlings (~2.5 cm per day; Mahoney and Rood 1998, Rood et al. 2005). Knowledge of when cottonwood disperse seeds and the conditions under which some of the resulting seedlings are likely to survive has been formalized as a conceptual model that was designed to guide managers in structuring flow releases: the recruitment box model (Figure 3). The model integrates the timing of seed release, the availability of suitable habitat, and the hydrologic requirements of seedlings after they germinate to help guide timing of peak flows and determining appropriate rates of stage decline. This model could easily be transferred to other species and applied to other hydrograph components. Site conditions of course determine how vulnerable a seedling is to desiccation, but the recruitment box model provides a general approach that can be customized to a particular species and a particular site to inform hydrograph design to accommodate cottonwood recruitment (Figure 3). The model has been applied to prescriptive management of flows in large rivers in the western US and in Canada with tremendous success in increasing cottonwood recruitment (Rood et al. 2005).

Years following successful cottonwood seedling establishment must be sufficiently high to provide moist soils for continued root growth, but can not be high enough to scour or bury seedlings. With time, the seedlings become increasingly resistant to scour, less dependent upon minor fluctuations in the water table, and able to sprout and adventitiously root in response to burial.

Because tamarisk releases seed later in the season and over a longer period of time than cottonwood, delayed peak flows could provide favorable conditions for tamarisk establishment. In addition to being used to enhance recruitment of desirable species, the recruitment box concept can also be used to inhibit undesirable species from becoming established.

Variability in flow from year to year plays an important role in maintenance of species rich plant communities relative to an area with ample moisture but a stabilized hydrograph (Figure 4). On any given year, the flow regime may facilitate recruitment and establishment of one species, while excluding another. Natural variations in the hydrograph include flood, drought, and variously timed events. Developing an understanding of which events support desirable or targeted species and those that hinder undesirable species helps to inform management decisions to achieve desired goals.

Because flow regime is a principle driver of riparian plant community structure, considerations of the magnitude of the peak, the level and duration of low flows, and the timing and rate of change in flow can assist in understanding the trade offs of various managed flows and help to refine how we use flow to meet specific management objectives.

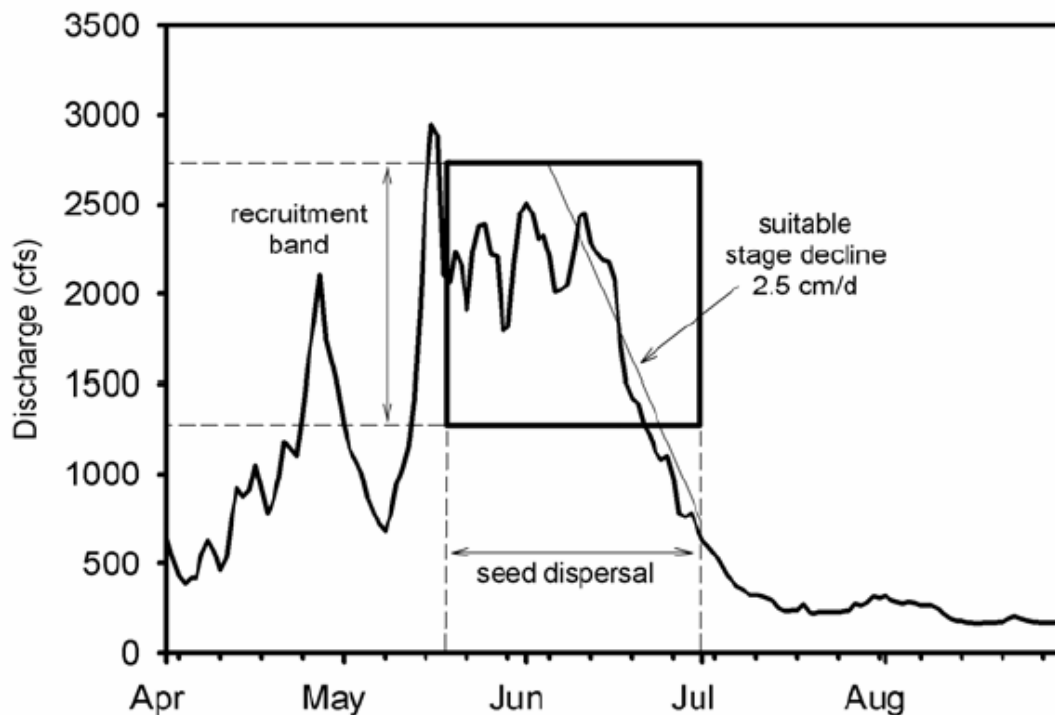


Figure 3. The recruitment box model illustrates the period of time over which cottonwood seedlings are most likely to survive and provides guidelines for the timing and rate of drawdown (the convention is stage decline, however discharge is shown here) for optimal cottonwood recruitment. Such guidelines could be used only when excess water is available. Hydrograph from Dolores River at Dolores April through August 1978 (Modified from Mahoney and Rood 1998).

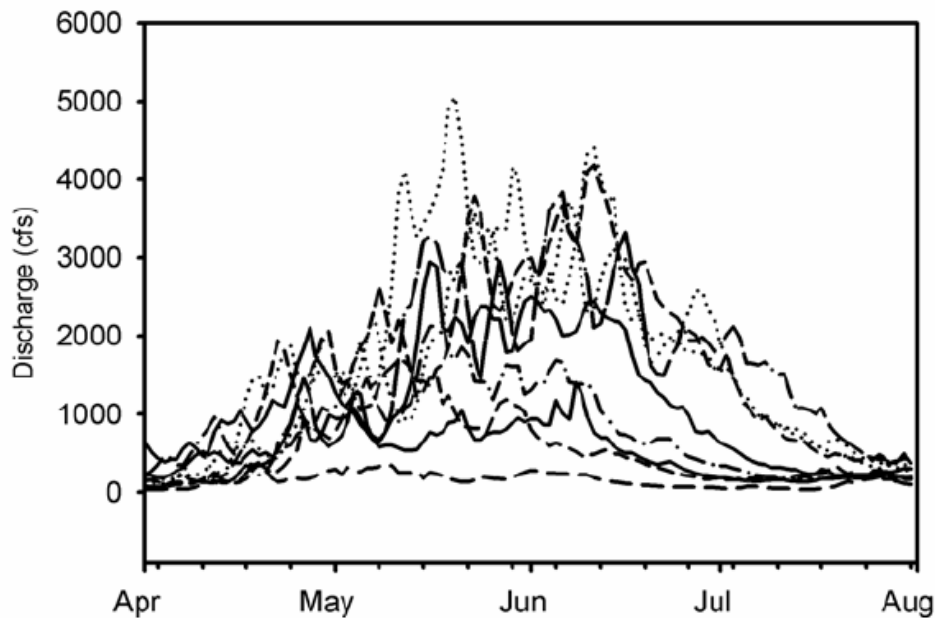


figure 4. Hydrographs of a variety of years on the Dolores River at Dolores illustrating the range of variation from year to year. It is this variability that favors some species and inhibits others on the same year, maintaining heterogeneity and species rich riparian plant communities.

Riparian vegetation of the Dolores River

Vegetation along the Dolores River downstream from McPhee Dam is typical of streams transitional between lower montane and Colorado Plateau desert environments. Woody riparian vegetation is characterized by sandbar willow, river birch, box elder, hawthorne, skunkbrush, red-osier dogwood, wild privet, and (in less confined reaches) narrowleaf and plains cottonwood and strappleaf willow. Herbaceous riparian vegetation includes three-square, spikerush, horsetail, red top, wild rye, saltgrass, Canada wild-rye, meadow foxtail, common-reed and a number of other forbs, grasses, sedges and rush species. Higher river terraces along some reaches include Ponderosa pine, Gambel oak, rabbit brush, big sage, and a range of other relatively drought tolerant woody as well as herbaceous species. Upland vegetation is dominated by Pinyon pine, western and Utah juniper, Gambel oak, sage, and saltbrush.

There are few published studies characterizing the vegetation of the Dolores River. Because riparian plant inventories have consisted of relatively coarse assessments or have focused on areas of small spatial extent, it is difficult to draw strong conclusions from existing data. However, the few reports and studies that do address riparian vegetation allow some general conclusions to be drawn.

The National Park Service conducted a wild and scenic river study on the Dolores and made some general conclusions about vegetation along the Dolores River in the late 1970s (USDI, 1979). According to these U.S. Department of Interior reports, vegetation along the reach of the

Dolores River downstream from Bradfield Bridge was in a “near natural” condition, being dominated by sandbar willow and extensive groves of cottonwood in the late 1970s. At the time, tamarisk was abundant downstream of Disappointment Creek and became increasingly dominant downstream (USDI 1979). The report indicated that there were extensive groves of cottonwood in the Gypsum Valley in the late 1970s. There are only a few old cottonwood in the valley today, but tamarisk is abundant, and monotypic stands are extensive (photos and field visits). Narrow canyon reaches of the Dolores River are not well-suited to support extensive cottonwood forests and are more likely to support shrub and dwarf tree communities (birch, sandbar willow, box elder, etc.). Wider valley segments are more conducive to the formation of extensive cottonwood forests.

Tamarisk was abundant along the Dolores River downstream of Disappointment Creek by the late 1970s but was and remains less dominant upstream to McPhee reservoir. Tamarisk is limited by frost and is thought to become abundant only below an elevation of about 7,000 ft on the West Slope in Colorado. It is not surprising then that tamarisk is not dominant upstream of Bradfield Bridge, but becomes increasingly dominant in low lying alluvial reaches downstream such as in the Big Gypsum and Paradox Valleys and near the confluence with the Colorado River in Utah. Because saline soils favor tamarisk, areas with high levels of salt may result in tamarisk being at an advantage over some native species along some reaches of the Dolores River. Sediment supplied to the river is exceptionally concentrated with salt due to exposed Mancos shale formations in tributaries and along the Dolores River valley. Salt has been further concentrated in low lying areas due to evaporation of water from shallow water tables and subsequent concentration of salt in the upper soil horizons. The visible salt crusts in some areas are known to inhibit the germination of cottonwood seeds and the survival and growth of cottonwood seedlings (Shafroth et al. 1995). Occasional overbank flooding could serve to flush floodplain soils and improve conditions for cottonwood and willow establishment and maintenance. Highly saline soils can increase the vulnerability of poorly adapted plants to problems associated with moisture stress.

Kriegshauser and Sommers (2004) measured riparian vegetation along a reach of the Dolores River near Lone Dome from 1988 through 2001 and recorded long-term trends. They recorded a significant increase in sandbar willow cover and number, no significant change in narrowleaf cottonwood cover or number, and a decline in silverberry over the 14 years considered. They noted that streamside meadows appeared to have become dominated by xeric (dry adapted) species. They noted that meadows historically flooded at 8,000 cfs or more, had rarely been flooded since the installation of McPhee Dam in 1984.

The increase in sandbar willow noted by Kriegshauser and Sommers (2004) is not surprising as sandbar willow can spread vegetatively by root sprouts so are not as dependent on hydrograph attributes for asexual reproduction. Since peak flows have been reduced on the Dolores (by 48% at Bedrock, CO; 20% at Cisco, UT), populations of willow that might have been kept from encroaching on the channel by scouring annual flows have slowly colonized the formerly active channel. The encroachment of willow is a predictable response to the reduction in annual peak flow. Particularly along gaining reaches or reaches with stable baseflow. Sandbar willow provides valuable habitat and is

a desirable species along western streams, but encroachment on the channel has the negative effect of reducing channel capacity.

Reductions in the cover and number of silverberry could be related to increases in soil salinity associated with reductions in the magnitudes of peak flows. Silverberry is tolerant of occasional flooding and can resprout after fire or scouring floods. Silverberry is often located in areas that may be considered transitional between the lower more frequently flooded areas where sandbar willow occurs and higher drier areas dominated by big sage and Pinyon-Juniper.

The Colorado Natural Heritage Program inventoried seven sites along the mainstem of the Dolores River from McPhee reservoir to downstream of the confluence of the San Miguel River in the late 1990s. They identified more than 50 plant species, and seven different plant associations. Three of these associations are considered to be globally imperiled due to rarity. These associations include box elder/river birch, box elder-cottonwood/red-osier dogwood, and strapleaf willow. Other associations documented included three square bulrush, sandbar willow/mesic graminoid, and skunkbrush. Associations along tributaries of the Dolores River included spikerush, box elder/river birch, and silverberry. Although these samples are not likely representative, cottonwood was not dominant in any of the stands sampled on the mainstem of the Dolores River, whereas it was the dominant in six of seven sites sampled on the lower San Miguel River within 10-15 km of the confluence with the Dolores River. The San Miguel is undammed and is considered to be one of the few relatively free-flowing rivers in the Colorado River basin. The plant associations identified above could be used to identify species to manage for and assist in the establishment of management goals targeted to specific riparian species or communities.

Because these reports and studies occurred over several decades and along different segments of the Dolores River, it is difficult to draw an overall assessment of the current state of the riparian vegetation of the Dolores River from existing literature. It would appear that cottonwood forests are less frequent and less extensive along the Dolores River than they were historically, that silverberry may be declining along some reaches, and that tamarisk has increased in abundance and extent (as it has on many western rivers including the free-flowing San Miguel) and that sandbar willow is more abundant than it was historically. It is likely that perennial species are more abundant along the Dolores River as well and that obligate wetland species have increased in abundance since relative stabilization of flows in the decades following the construction of McPhee dam (Merritt and Cooper 2000).

Given this cursory assessment drawn from scant existing data, it becomes clear that a vegetation inventory of the Dolores River would provide invaluable baseline data on the current condition of the vegetation and would facilitate the tracking of trends in the condition of the vegetation through time. An assessment of current condition of riparian vegetation along the Dolores River would assist in setting goals for management and shed light on the processes that could be most efficiently manipulated to achieve some desired outcome. An inventory would also provide valuable information that could be used for setting realistic goals that account for management and physical constraints. It is likely that the current structure of the riparian vegetation along the Dolores River is an integration of the changes that have occurred not only in flow regime, but also in grazing, human use, and other factors.

An inventory would also help to expose ways to maximize the yield on the investment made in efforts to restore certain components of the riparian system. For example, well-established stands of tamarisk along channel margins would probably not respond negatively even to severe flooding. So if the desired goal of a spill were to enhance cottonwood regeneration, an understanding of the non flow-related constraints on cottonwood establishment could prompt some pre-spill investments. Mechanical removal of tamarisk followed by flooding could result in the erosion of banks and the creation of new sites for cottonwood establishment. Another advantage of having the information from an inventory is that the sites could be re-sampled following such activities to provide a quantification of the vegetative response to various activities (prescribed floods, changes in low flow, mechanical removal of undesirable species, etc.).

An approach to inventorying the riparian vegetation of the Dolores River

One important aspect of inventorying for the purpose of recording trends through time is to design a network of permanent plots that can be re-sampled at some set time interval or prior to and following management activities. Because riparian plant communities are organized around the geomorphologic features of the channel, floodplain and terraces and are a reflection of the fluvial processes that formed and continue to influence the features, a monitoring protocol should be designed to include a full range of geomorphic settings.

The reaches defined in the DRD geomorphic analysis section of this report grossly reflect valley-scale geomorphic patterns but include (in some cases) both confined and unconfined reaches. The reach of the *Dolores between McPhee Dam and the Bradfield Bridge* is a reach characterized by an unconfined river channel that historically meandered across the entire valley bottom and supported extensive cottonwood and willow forests. The reach of the Dolores from *Bradfield Bridge to the San Miguel River* is generally higher gradient than the McPhee Dam to Bradfield Bridge Reach, has a narrower valley bottom, and the channel is confined between steep canyon walls (with the exception of the reach through the Big Gypsum Valley). The Big Gypsum Valley is a wide valley with a lower gradient channel and finer textured bed and banks than the more confined, higher gradient reaches. The *San Miguel River to the Colorado River* reach of the Dolores varies considerably from narrow canyon (upstream from Gateway) to wide valley (downstream from Gateway) and canyon and alluvial reaches until the confluence of the Dolores and Colorado Rivers.

A hierarchical geomorphic classification of segments, reaches, and fluvial features would be a useful organizational tool for stratifying a vegetation sampling protocol for the Dolores River. For example, classifying *stream segments* based upon some combination of valley width (degree of confinement), channel gradient, and sinuosity could be the first level in the hierarchy and would likely result in two or three characteristic segment types. The next level in the hierarchy might describe channel form at the *reach scale* as it is influenced by colluvial deposits adjacent to or in the channel, alluvial fans formed at tributary junctions, and constrictions and expansions that influence hydraulics and scour and deposition. The third tier in the classification could be at

the level of the *fluvial feature* and may include such categories as terrace, active floodplain, side channels, islands, point bars, reattachment bars, etc. Because vegetation patterns along rivers are strongly associated with geomorphic features, a vegetation sampling scheme would be designed to systematically sample the range of geomorphic features identified in this classification.

One approach to sampling vegetation is to establish permanent cross-sections, perpendicular to the river channel at pre-determined intervals. Systematically placing transects so that the full range of fluvial environments is represented is another sound approach. The number and placement of cross-sections is usually determined by accessibility, resources, and time. Once cross-sections are located and permanently marked (i.e., with rebar end posts), herbaceous vegetation may be sampled in framed plots placed at regular or random intervals along transects (1-4 m² is an acceptable range of plot sizes). Alternatively, if time permits, plots may be laid out continuously (belt transects) along a measuring tape from the edge of low water to the landward extent of the riparian zone. Woody vegetation may be sampled using the line-intercept method with visual estimates or measurements using a densitometer.

Characterization of important abiotic variables may then be directly related to vegetation in the plots. For example, if rating curves were established for a site, plot specific flood frequency, inundation duration, etc. could be directly related to the elevation/stage of the plot. In addition, variables such as soil texture and chemistry could also be gathered. These abiotic factors could then be formally related to plant community composition in the plots. From this, causal explanations for the observed patterns in plant community composition, age-class structure, and the species or life-history guilds might become more apparent. In addition, our confidence in predictions about the probable response of riparian vegetation to changes in driving variables (such as flow-related factors) would increase with such information (Auble et al. 2005).

Understanding the patterns that currently exist along the Dolores River is a fundamental first step in making flow recommendations. Once a baseline of plant species composition data measured at permanently marked sites is established, the inventories could be repeated annually or after extreme events (floods or droughts) to refine our understanding of cause and effect along the Dolores River.

Changes in flow regime on the Dolores River, flow components, and vegetation

Along the Dolores River there have been many biologically relevant changes in flow regime as a direct result of the operations of McPhee dam. The following text does not attempt to assess whether these changes are positive or negative, but rather points out the biological responses to changes in physical factors such as hydrologic and fluvial processes.

Indicators of hydrologic alteration analysis was conducted on the Dolores River at Bedrock, CO (USGS 09169500) and at the Dolores River near Cisco, UT (USGS 09180000) gages. Summary statistics were calculated to compare the daily average flow records for pre and post dam periods of time. Flood frequency curves were also fitted to annual instantaneous peak flow data using the log Pearson type III distribution. The magnitudes of recurrence interval flows of interest were estimated from these flood frequency curves.

High flows

Peak flow and the average monthly flow for the high flow months were all reduced considerably in comparisons of the pre to post dam periods. The average instantaneous peak flow at Dolores River at Bedrock for the period 1918-1983 (non continuous record) was 5,023 cfs and the post dam (1984-2004) annual average was 2,595 cfs (-48%). The maximum flow of record (9,280 cfs in 1973) during the pre dam period was nearly double the average peak. The lowest instantaneous peak flow during the pre dam record at the Bedrock gage was 1,290 cfs (in 1981), whereas the lowest peak during the post dam period was an order of magnitude lower; 164 cfs. Average instantaneous peak for the Dolores River near Cisco, UT gage for the pre dam period 1951-83 was 6,900 cfs and the post dam period (1984-2004) was 5,486 cfs. The maximum flow of record --17,400 cfs recorded in 1958-- during the pre dam period was 2.5 times greater than the pre dam average peak. The lowest instantaneous peak flow during the pre dam record at the Cisco, UT gage was 2,110 cfs (in 1981), whereas the lowest peak during the post dam period was 1,260 cfs (in 1996).

The snowmelt peak along the Dolores River for the period of record typically occurs before the end of June and monsoon peaks may occur from July through September. The average date of the snowmelt peak at Bedrock is 12 May. Average flow during the month of May decreased by 40% at Bedrock but stayed about the same (+1.5%) near Cisco, UT. Average June flow in the Dolores decreased by 54% and 18% at Bedrock, CO and near Cisco, UT, respectively. The 10 year recurrence interval flood has decreased from 9,040 cfs to 5,500 cfs (-39%) at Bedrock, CO and from 13,500 cfs to 10,000 cfs (-26%) near Cisco, UT since McPhee reservoir operations began.

The 48% and 20% reductions in instantaneous peak flow in the Dolores River at Bedrock, CO and Cisco, UT have likely resulted in shifts in plant community composition and encroachment of vegetation along a significant segment of the Dolores River. As described in the “River processes, riparian vegetation and flow components” section above, the reduction in the magnitude of peak flow reduces the frequency and extent of inundation of the floodplain resulting in decreased microbial activity and potentially to increased concentrations of salt in floodplain soils. Reduction in the frequency of the high flows that mobilize and transport material, rework channel deposits, scour and remove vegetation from channel margins, and transport and deliver seeds and propagules of plants across the floodplain all contribute to the changes in plant community structure that have likely occurred along the Dolores River.

The annual floods have not been of sufficient magnitude to maintain channel margins free of perennial plants (as noted both in the “DRD geomorphology analysis” section of this report and in Kriegshauser and Sommers 2004). These smaller annual floods have likely resulted in a reduction in the turnover rate of marginal bars, point bars and medial bars and islands, resulting in stable channel features relative to pre dam conditions. As a result of this relative stability, channel margins, point bars, and medial bars likely have higher cover of vegetation and probably a higher proportion of perennial species than would have occurred during the pre dam period. It is likely that disturbance adapted species, such as narrowleaf cottonwood, have decreased in frequency along the alluvial reaches of the Dolores River as a result of the factors associated with reductions in the magnitude of high flows.

Cottonwood recruitment generally occurs on or in years following high flow events. Some have generalized that recruitment of cottonwood is tied to the 10 year recurrence interval flow (Scott et al. 1997). Along the Dolores the 10 year recurrence interval flood has decreased by 26 to 39%. Prior to the operation of McPhee, the 10 year recurrence interval flow was 9,040 cfs; such a flow has not occurred along the reaches represented by the Bedrock, CO gage since the reservoir went into operation. Age-class distributions of the cottonwood populations along the Dolores River today are likely to reflect the decreased magnitude of the 10 year recurrence interval flow. Anecdotal observations from Paradox Valley and Big Gypsum Valley confirm that there are predominately senescing stands of cottonwood and very few younger age-classes (D. Graf, pers. comm). The highest flows since construction of McPhee have been in the late 1980s and in 1993 and none have exceeded the 5,230 cfs peak that occurred in 1986. On years of particularly high snowpack it would be beneficial to disturbance-adapted riparian species to release appropriately timed, high magnitude (i.e., overbank) flows from McPhee dam.

Low flows

Low flows (the 1, 3, 7 and 30 day minimums) have increased along the Dolores River at Bedrock, CO by from 230% to 590%, from flows less than 10 cfs to flows over 30 cfs. The pattern is similar along the Dolores near Cisco, UT. The 1, 3, 7, and 30 day inimum flows have increased by 66% to 106%. The stability and reliability of these low flows has likely resulted in an increase in hydrophytic vegetation along channel margins. Species such as three square, rushes, sedges, brooklime, fieldmint and other water-loving species have probably increased along the margins of the Dolores River. The fluvial wetlands that have likely increased in extent along the Dolores over the past couple of decades are likely species rich, productive, and provide valuable forage and habitat for insects, birds and mammals. Whether the values of these newly formed wetlands rival the values lost from the observed decrease in the extent of cottonwood forest is an area for future investigations.

The increase in baseflow (in conjunction with decreases in high flows) has likely facilitated the encroachment of sandbar willow into the channel as well. The herbaceous wetland communities can withstand prolonged periods of inundation but rely on a relatively stable water table near the channel during periods of low flow. Since surface and groundwater are tightly linked along alluvial reaches of the Dolores, elevated baseflow is likely reducing the occurrence of water stress in phreatophytes positioned further from the channel (eg., cottonwood, tamarisk, willow).

If low flows were modified to emulate pre-McPhee conditions, Phreatophytic vegetation could become water stressed during hot, dry periods and wetland communities along channel margins would probably either shift closer to the channel over time or be replaced by more drought tolerant species. Unless the high flows are restored as well, it is inadvisable to reduce the low flows as this would cause reductions in the extent of the wetland communities formed in the post-dam period.

Timing and rate of change in flows

The timing of the snowmelt peak has shifted along reaches represented by both the Dolores River at Bedrock, CO and near Cisco, UT gages. The snowmelt peak has historically occurred between April 13 and May 28 at the Bedrock, CO gage (median May 18). Since the dam has been in operation, the peak has occurred between April 9 and June 22 (median May 22). Most of

the highest peak flows on record since McPhee went into operation have been inappropriately timed for coinciding with period of cottonwood seed release (late May early June). Instantaneous peak flow exceeded 4,000 cfs in 1984, 1985, 1986, 1987, and in 1993. The timing of the peak was April on all years except in 1986 and 1987, which is prior to the period of seed release for cottonwood along the Dolores (late May-early June). It is likely some cottonwood cohorts became established along the Dolores during 1986-87 flooding of 5,230 cfs on May 5 and 4,390 cfs on May 21, but these cottonwood would have become established at lower elevations on the floodplain than they might have in a pre-dam recruitment year.

It is advisable to time the peak within the window of historic peak flows. The median pre-dam date is 18 May provides an ecologically meaningful target that likely relates not only to cottonwood and other flow-adapted native plant species, but also to native fish, amphibians, and invertebrates (Lytle and Poff 2004). The general guideline for structuring the rate of drawdown following the peak is that the rate of stage decline should not exceed about 2.5 cm per day during and after the period of cottonwood seed release. This 2.5 cm per day has been shown to be a rate corresponding to the maximum rate of root growth for cottonwood (*Populus deltoides*) and is considered optimal for cottonwood growing in medium to coarse sand (Rood et al. 2005). In finer sediments, soil water holding capacity is higher and rates of drawdown could be increased. In areas with coarse-textured substrate, the further rates of drawdown deviate from this optimal rate, the higher the probability of failed cottonwood recruitment. This stage decline should be calculated for those areas in which cottonwood recruitment is most likely (i.e., areas where bare patches of alluvium have been formed).

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Tamarisk Summary

Review of an Invasive specie

By Jim Siscoe

Exotic species are often classified as invasive or weedy plants. The dynamics of biological invasion suggest that it may be useful to study habitat requirements and physiology of both exotic and native species. In this chapter I will discuss how for many plants species the interaction of plant physiology and the environment influences population dynamics.

Invasion of alien species into new regions is an important issue in biology today (di Castri et al., 1990). Typically the invasion of a species into an area is characterized by its rapid spread over large areas (Hengeveld, 1989). This spread may show two kinds of dispersal, a steady advance of a population or the scattering of satellite populations from a central location (Baker, 1986). Areas usually thought of as vulnerable to invasion include grasslands, riparian habitats, waterways, roadsides, and trodden paths (Baker, 1986). Many species are known to invade these types of habitats with great success. Gouyon (1990) describes an invasive species as one which “is not at equilibrium from an ecological and/or demographic point of view.” He claims that to understand what an invader is means understanding what the mechanisms are that maintain equilibrium. Riparian areas are habitat types which rarely are in equilibrium. Therefore to study the dynamics interactions between the abiotic community and the biotic community can be very difficult.

Riparian areas are typically referred to as woodlands which are adjacent to and dependent upon, water resources which are permanent (such as creeks, streams, and rivers), or intermittent flows such as arroyos (Chaney et al., 1990). A General Accounting Office (1988) report refers to riparian areas as “narrow bands of green vegetation along the banks of rivers and streams.” Riparian areas can be very complex and have a unique role in the arid and semiarid southwest (Kauffman and Krueger, 1984). The biological importance of these areas to humans and the surrounding wildlife is out of proportion with the relatively small area they comprise (Szaro, 1989). Although these areas make up only about 1 percent of the land in the Western United States, they are probably the most productive and diverse sites in the southwest and thus have received much attention in recent years (Reichenbacher, 1984; Chaney et al., 1990).

An important factor which shapes many southwestern riparian habitats is disturbance. Many riparian species respond to two important factors: (1) an unstable substrate and (2) variable available soil moisture. Hummel (2003) describes one of the major reasons for changes in the disturbance regime along many southwestern riparian areas today as a result of man made dams and diversions which have changed the flow of most major rivers in the United States. He indicates that alterations in river flow have affected the rate and frequency of flooding, sediment discharge, and nutrient transport. As a result the establishment of native seedlings is diminished. He also states that “We have a compounded problem if the opportunistic species is a non-native, invasive species such as *Tamarix*”. This is especially true in the southwest where abundant soil moisture and the change in flow frequency from early spring flows to low or NO FLOWS to a perceived natural hydrograph and perennial flows below dams.

(Reichenbacher, 1984). Disturbance such as flooding is a frequent event and is an important abiotic factor in many riparian zones (Reily and Johnson, 1982; Gecy and Wilson, 1990; Ripple,

1990). Changes which occur in floodplains due to flooding are complex, leading to changes in stream channel width, sinuosity, and vegetation patterns (Szaro, 1989). Although flooding is widely recognized as a key disturbance determining vegetation composition of riparian areas, intensity of flooding is just as important as frequency (Odum, 1978). Although flooding is important it may not be the only key factor shaping the composition of southwestern riparian communities today. Along many of the riparian areas in the southwest today, salinity may also be as important as flooding when influencing the ecology of many riparian systems in the southwest.

Annual flooding is the primary disturbance that occurs in southwestern riparian areas and many native riparian trees such as cottonwoods (*Populus* spp.) and shrubs (*Salix* spp.) have adapted to spring flooding caused by snow melt (Szaro, 1989; Howe and Knopf, 1991). However, many waterways of the southwest have now been altered because of dam building which has effected the intensity and frequency of annual floods. Many species found along southwestern riparian habitats today are dependant on flooding to provide sediment deposition along the banks for the establishment of new seedlings. Many of native riparian species found in the southwest produce an abundance of seeds which are extremely small and in the case of cottonwoods are accompanied by “cottony enclosures” (Rood and Mahoney, 1990).

The seeds of these species and many invasive species are viable for a short time and are usually wind-dispersed during May and early June (Rood and Heinze-Milne, 1988; Rood and Mahoney, 1990; Siegel and Brock, 1990; Howe and Knopf, 1991). Important for good seed germination and survival is the establishment of open silt beds which form during floods. Seeds will not germinate unless there is adequate moisture and sunlight and survival depends primarily on water resource availability. Thus, southwestern riparian zones are ideal for invasive species to become established. This is primarily due to seasonal disturbance which occurs in these areas creating an environment for seeds of invasive species to become established which also have similar germination requirements. However, as I will discuss later in this paper this is not the only reason for their success.

Many invading species are characterized by a diversity of ecological, physiological, genetic, and morphological characteristics (di Castri et al., 1990). Baker (1986) uses the phrase “ideal weed” to summarize the traits often found in invading species. An ideal weed is classified as “a plastic perennial which will germinate in a wide range of physical conditions, grows quickly, flowers early, is self compatible produces many seeds which disperse widely, reproduces vegetatively and is a good competitor.” Drake et al. (1989) is careful to point out that although a particular invasive plant may possess one or many of these characteristics success within a given community is not guaranteed.

Many of the invasive plant species found along southwestern riparian streams demonstrate one or all of these characteristics and are generally obligate hlophytes that are confined to areas of low-- to moderate salinity. The most common are spp. (saltcedar) (Russian Olive. Both have been shown to have considerable advantage over native species by exploiting suitable habitat for germination over a longer time interval (Howe and Knopf, 1991). Generally the advantages are, that these species have also exhibited a greater ability to cope with changes in water flows and salinity. The habitat requirements of these species will also be compared to the habitat needs of

(cottonwoods) and spp., (willow) which are both members of the family Salicaceae and are native to southwestern riparian areas.

Description of Tamarisk

Found in the family Tamaricaceae there are about 75 species of Tamarix which are very similar to one another (Siscoe, 1992). Other studies have claimed there are only 53 species of Tamarix (DeLoach, et al., 2000). Tamarix is a deciduous or evergreen feathery tree or shrub. The flowers are bisexual with 4 to 5 petals and sepals, 4 to 5 stamens, and 3 to 4 stigmas (Clapham et al., 1962; Waisel, 1972; Elmore, 1976). Tamarix or saltcedar, according to some authors, was first introduced into the U.S. from the Mid-East during the late 1800's as an ornamental, and to prevent erosion along stream banks (Ungar 1974; Hoddenbach 1987; Davila 1989; Everitt and DeLoach, 1990). One author, however, speculates that saltcedar may have been introduced as early as the late 1500's by Spanish explorers. This idea is rejected by a 1977 draft E.I.S. report stating that "if saltcedar were introduced at this time, it would be much more abundant in Mexico than it is" (Dept. of Int., 1977).

The Tamarix species found throughout the southwest are native to the Mid—East, Africa, Asia, and possibly the Mediterranean (Elmore, 1976; Everitt and DeLoach, 1990). With the exception of some areas in China Tamarix in its native habitat is restricted to hot environments with high soil and water salinity levels. Tamarix species exhibit morphological characteristics such as small succulent-like leaves, sunken stomata, water storing tissue, and salt glands which enable these species to live in a highly xeric environment (Ungar, 1974). The three primary species of Tamarix found throughout the southwest are Tamarix chinensis Lour., Tamarix ramosissima Ledeb., and Tamarix pentandra Pallas. (Note: T. pentandra is considered to be the same species as T. ramosissima (Baum, 1978)). All of these species form low dense thickets along riparian corridors displacing native vegetation, impeding water flow and increasing soil salinity (Everitt and DeLoach, 1990). *Some have argued that the taxonomic status of Tamarix chinensis Loureiro may need to be changed.* This species is commonly known as tamarisk or saltcedar. A microscope, or at minimum a hand lens, are required for the proper identification of any species of tamarisk. Ladyman, (1991), states that "Tamarix pentandra var. pentandra Pallas has been mistakenly applied to this species within North America. Actually, T. pentandra Pallas is synonymous with T. hispida Willd. that is distributed throughout central Asia". He also states that confusion has resulted from this species being referred to as a sub-species of T. gallica L., which is a unique taxon originally from Europe, and now also naturalized within the United States .

Tamarisk effects on the local ecology

Soil salinity

Western rivers and streams flow through areas of naturally saline soils and groundwater. Spring flooding can leach out these salts but with the present pattern of no flooding or of only low floods, the salts continue to accumulate. Tamarisk has been shown to be much more salt tolerant than are cottonwoods, willows, and most other vegetation along many southwestern riparian habitats (DeLoach, et al., 2002). Because of this advantage tamarisk has a competitive advantage over many other vegetative species considered to be native to southwestern riparian habitat types.

DeLoach, et.al, 2002 reported that Tamarisk creates “a feed-forward process in which it invades, concentrates salt in its foliage which then drips to the soil surface or falls with the foliage, which forms a saline litter layer and surface soil, which favors SC seedlings over native salt intolerant species, thus further accelerating SC dominance, which causes even more surface and soil salinization, etc”. This ability of Tamarisk is a result of its physiological characteristic, having “salt glands” in its leaves which allow tamarisk to uptake high concentrations of salts, dump these salts to the leaves and then shed these leaves when the concentration of salts in the leaves becomes too high (Siscoe, 1992).

Within the study area one of the areas most affected by this feed back loop is Reach 6 and to a lesser degree Reach 7.

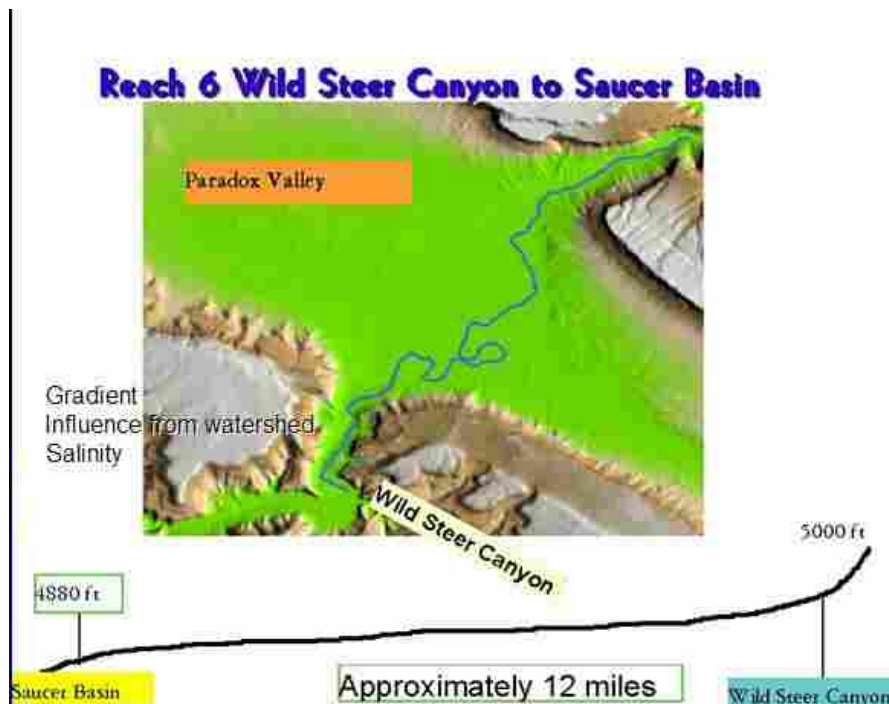


Figure 1 (Map showing reach 6 of study area)

Another factor in this “feed back loop” comes from the transpiration rate of tamarisk. A). Hummel (2003) states that, “although all plants transpire, some transpire more than others. Salt Cedar transpires at the same rate as the native trees during non stressed conditions, but during stressed conditions (high salinity or low water) the Salt Cedar maintains higher levels of evapotranspiration”. This has the affect of reducing available water for plant uptake and thus compounds the salinity problem.

Tamarisk Habitat Studies:

Saltcedar is found primarily along rivers and streams, and can also occur along roadsides, old drainage ditches and stock ponds. Riparian habitats dominated by tamarisk are some of the most common vegetative communities in the southwest. The tamarisk species discussed in this paper have been shown to grow from elevations ranging from 160 m to 2150 m (Hoddenbach, 1987). A cluster analysis of 153 tree and shrub communities found along riparian areas of Arizona and

New Mexico found that Tamarix pentandra occurred in only one site below 1000 m. In another study conducted along the Pecos River tamarisk behavior was illustrated by its colonization of the Pecos River Valley in New Mexico. In 1912, a few seedlings were observed at Lake McMillan. By 1915 the shrub covered 600 acres and extended up and down the riverbanks. By 1926 it covered 12,300 acres and extended to 57,000 acres by 1960. It out-competes many native species and is difficult to eradicate once established (DeLoach,et.al, 2002). Hummel (2003) found that *Tamarix* decreased the richness and abundance of aquatic microinvertebrates and this reduction may affect the higher trophic levels negatively. He states in his paper that “The leaves of Salt Cedar can contain up to 15% soluble salts which may render it useless to larger herbivores while on the tree and both above or below the water surface once the leaves are shed”. He also, found that *Tamarix* leaves are a poor food source for arthropods because of the level of tannins in their leaves.

SUMMARY

In many parts of the southwest riparian areas have been influenced by a variety of events. A variety of issues in the southwest such as ; Dam building, cattle grazing, and the introduction of exotic plants have created vastly different habitats in these areas than previously existed according to many individuals. Although dam building and grazing have impacted riparian areas of the southwest, they have not had the kind of ecological effect on riparian areas of the southwest that Tamarix has shown. The invasion of tamarisk into many riparian areas of the southwest has led to decreased habitat for native fauna, displacement of native shrubs, and the probable increase in soil salinity, which in turn creates an even better habitat for tamarisk. Few studies have conducted true experiments in order to understand more fully the influences of tamarisk on the overall ecology of riparian areas of the southwest. There is considerable debate in some circles of government land agencies about the usefulness or the feasibility of trying to control tamarisk in the southwest. However, this is changing and the Dolores River Dialogue may offer a unique opportunity to study the effects of tamarisk removal on the ecology of key habitat areas of the Dolores River.

Potential flow recommendations and habitat enhancement

Hummel (2003), suggests that alternating between “a first year short duration scouring summer flood to remove some of the *Tamarix* near the channel, but then keep the flow rate low to reduce the summer germination of *Tamarix* away from the channel. The following year go back to a treatment with no spring flooding and then a large summer flood”. He believes that by monitoring the germination of seedlings, “one may be able to determine the optimal time for the survival of the native species or the optimum time for the mortality of the invader and control the releases and the intensity of the releases accordingly”. Although I would not disagree with Hummel’s assessment of alternating flows to monitor seeding survival, I would suggest the following issues must also be considered;

1. Flow management alone may not remove enough tamarisk from the channel to provide needed habitat improvement. Therefore, mechanical treatment would be necessary to see real tangible results.
2. Recognize that although tamarisk produces an abundance of seeds most tamarisk found throughout the study area reproduces vegetatively.

Ultimately the rehabilitation of the channel and adjacent riparian area will likely be dependant on a coordinated effort between flow management and mechanical habitat improvement in “key” habitat types within the study area.

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Appendix A

Focus Questions

▪ **Focus Questions and Core Members assigned to each task**

Key--- Lit= Literature review (published as well as unpublished)

GIS= Baseline mapping and some hydrology analysis

FW= On site visits (obviously limited to number of visits and sites due to time constraints)

Cold Water Sport Fishery

QUESTIONS

- What flows create and maintain habitats that support life cycles of cold water sport fish (e.g., salmonids)? Lit
- How valid is the characterization of ‘cold water habitat’ for reach 1 between the McPhee outlet and the Bradfield Bridge? Lit, Fw, Gis
- For cold water fish in reach 1 (as a working assumption), can mechanical habitat improvement projects be cost effective in improving downstream salmonid habitat? Lit, Fw, Gis
- Is there evidence that previous habitat improvement projects were successful? Lit
- Given the Hydrologic Summary Report (“Opportunities and Constraints”), has post-project perennial flow affected the habitat for cold-water sport fish below McPhee? Lit, Fw
- Given the Hydrologic Summary Report (“Opportunities and Constraints”), how might flow management be used to positively affect habitats for cold water fish species? Lit, Gis
- Are these flows supportive for both guilds of species, or are they in conflict within other reaches of the study area? Lit
 - Magnitude
 - Duration
 - Recurrence intervals
 - Timing
- Is there evidence of predation on cold-water sport fish, including terrestrial species, by otters or other species? Lit, Fw
- Given operational parameters re: use of the selective outlet works (stratified release of McPhee water), what, if any, are the implications to the sport fishery below McPhee if non-native warm water sport fish (e.g., smallmouth bass, crappie, sunfish...) are inadvertently released downstream? Lit
- Are there opportunities to enhance cold water fisheries in other areas of the study area beyond reach 1? Lit, Fw, Gis

Native Fishery (Gigi Richard, David Graf, Rick Anderson Jim Siscoe)

QUESTIONS:

- What flows create and maintain habitats that support life cycles of native warm water fish? Lit
- What are the life cycle limiting factors for native warm water species? Lit
- Is there any data on primary aquatic production (benthic or other macro-invertebrate populations) that could aid in assessing the habitat/ life-cycle requirements for native species? Lit
- Any possibility to use the benthic community as a metric or bio-indicator for the health of the native aquatic community? Lit
- Where, specifically, do “warm water”, “cold water”, and “transitional” habitat types currently exist? Lit, Fw, Gis
- Where could they exist? Lit, Fw, Gis
- How do they vary temporally? Gis
- How could they vary given more optimal flow management favoring native aquatic species? Lit, Gis
- Are there options to use mechanical habitat manipulations (cost effectively) for improvement of habitat types for native species? Lit, Gis
- Given the Hydrologic Summary Report (“Opportunities and Constraints”), have post-project perennial flows affected the native fisheries habitats below McPhee? Lit, Fw
- Given the Hydrologic Summary Report (“Opportunities and Constraints”), how might flow management be used in the future to positively affect habitats for native fish species? Lit
 - Magnitude
 - Duration
 - Recurrence intervals
 - Timing
- Given operational parameters re: use of the selective outlet works (stratified release of McPhee water), what are the implications to native fish species if non-native warm water sport fish (e.g., smallmouth bass, crappie, sunfish...) are inadvertently released downstream? Lit
- Which release levels create the greatest risk to downstream native species? Lit
- Is there evidence of predation on native species, including terrestrial species, by other non-native aquatic species, or otters? Lit, Fw
- What is the status of the mottled sculpin, the only cold water native species below McPhee? Are there flow-related risks to this species? Lit, Fw
- Where have native fish populations been most abundant, pre McPhee Dam? Lit, Fw

Geomorphology (Gigi Richard, Andrew Wilcox, Jim Siscoe)

QUESTIONS:

- What process will support the geomorphic processes necessary to create and maintain desirable habitat components? Lit
 - What is the current geomorphic character of the river? (Reach scale classification – current and historic context)
 - In the post-dam era, what is the channel forming (e.g., bankfull) flow?
 - To what extent has channel adjustment in alluvial reaches occurred since dam closure?
 - Are there human constraints, beyond the dam to channel forming process?
 - What are the most important pre- and post-dam geomorphic processes affected by dam closure?
 - What are the potential future condition relative to habitat components identified by other resource disciplines?
 - Are there different opportunities for these components to be supported by the (“Opportunities and Constraints” hydrology within the study area ?
 - What flows (timing, duration, magnitude, recurrence intervals) maintain pool scour, cleanse and sort spawning gravel, and flush fines and other introduced sediments?
- What is the current geomorphic character of the river? (Reach scale classification? Gis
- Can flows (timing, duration, magnitude, recurrence intervals) maintain pool scour, cleanse and sort spawning gravel, and flush fines and other introduced sediments given a specific reach gradient and constraint (i.e. are roads and other human influences affecting the rivers natural course? Lit, Fw, Gis
- What is the potential of tributary sediments and hydrology introduced below McPhee Dam for altering channel morphology and providing habitat improvements for fisheries and potential riparian plant species? Lit ,Gis
- Is sediment loads introduced below McPhee Dam from other tributaries having an affect on water chemistry, specifically salinity levels? Lit, Fw, Gis
- What flows and gradients will support the geomorphic processes necessary to create and maintain potential habitat components for fisheries and riparian plant communities? Lit

Riparian Ecology (Dave Merritt, Preston Somers, Andrew Wilcox, Jim Siscoe)

QUESTIONS:

- What process create and maintain the native riparian communities in the Lower Dolores River corridor? Lit
- What flows can affect changes in the riparian communities that support age class distribution or discourage native riparian vegetation given current conditions? Lit, Gis
- Can flow management be an effective tool in selecting for native over non-native riparian species? Lit, Gis
- What is the current distribution of riparian vegetation along the Dolores River, below McPhee Dam? Gis
- Have historic flows created and maintained native riparian communities? Lit, Gis
- Are there human constraints to maintaining and or improving the habitat for native riparian plant species? Lit, Gis
- Are there reaches along the study area that are more suited to specific riparian habitat types than others? Lit, Fw, Gis
- What are the potential riparian habitat types within each reach? Lit, Gis
- Is there evidence that climate is altering the ability of some riparian species to reproduce and mature, regardless of normal flow patterns? Lit
- Can the dissolved organic carbon budget be addressed within the study area? Lit
- Is salinity affecting the riparian habitat within the study area? Lit, Gis

Tamarisk

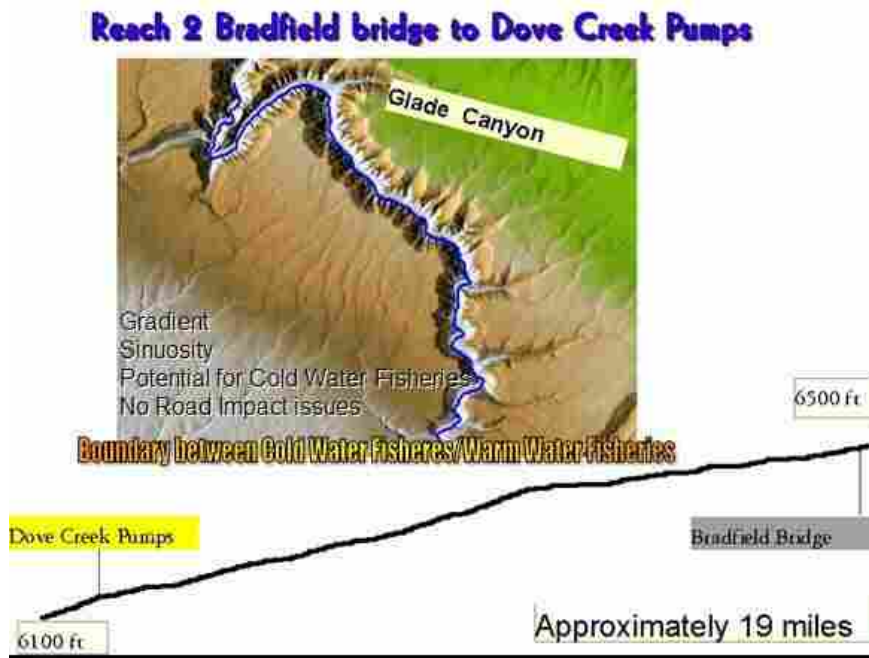
- What is the role of tamarisk in the lower river ecosystem? Lit
- What is the distribution of tamarisk along the Dolores River below McPhee Dam? Gis
- What is the effect of tamarisk on native riparian tree ,shrub and macro invertebrate species along the Dolores River below McPhee Dam? Lit
- Has perennial flows increased the opportunity for tamarisk to expand its habitat? Lit

Appendix B

(Note, elevations and miles shown in this appendix are approximate)

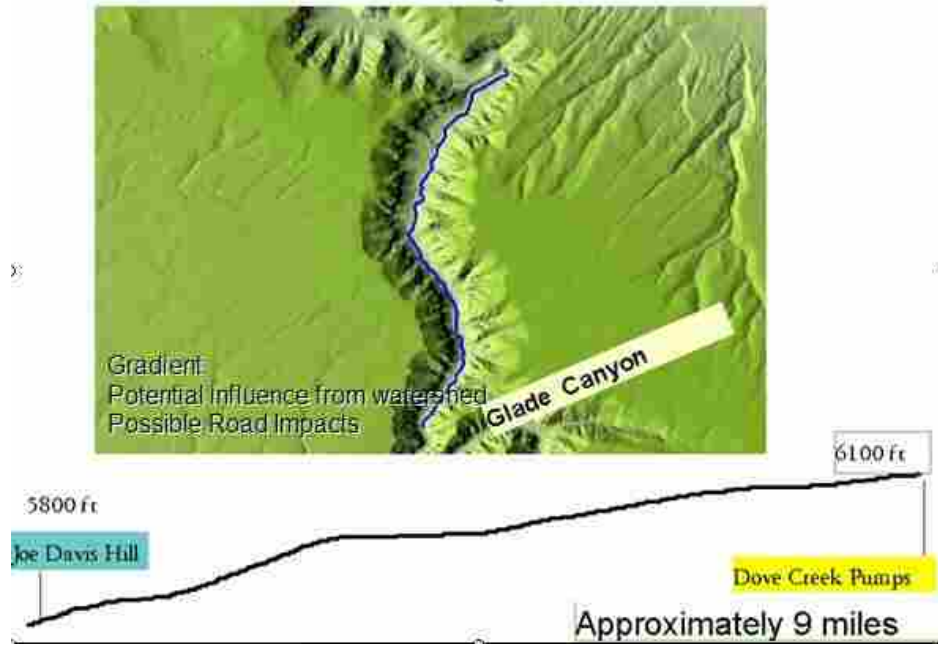


Reach 1 location



Reach 2 location

Reach 3 Dove Creek Pumps to Joe Davis Hill



Reach 3 location

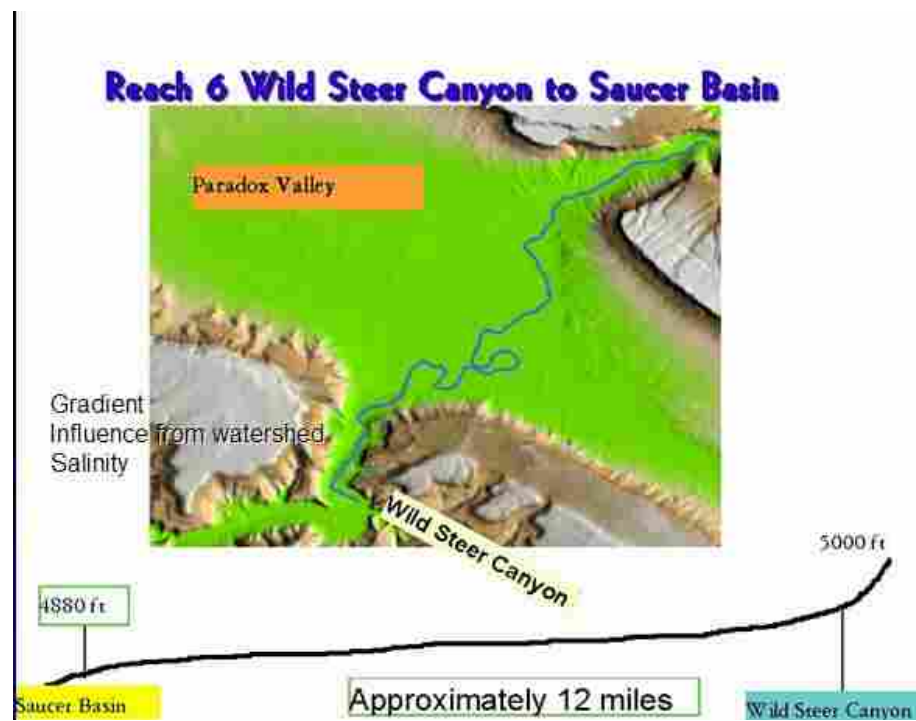
Reach 4 Joe Davis Hill to Big Gyp



Reach 4 location



Reach 5 location



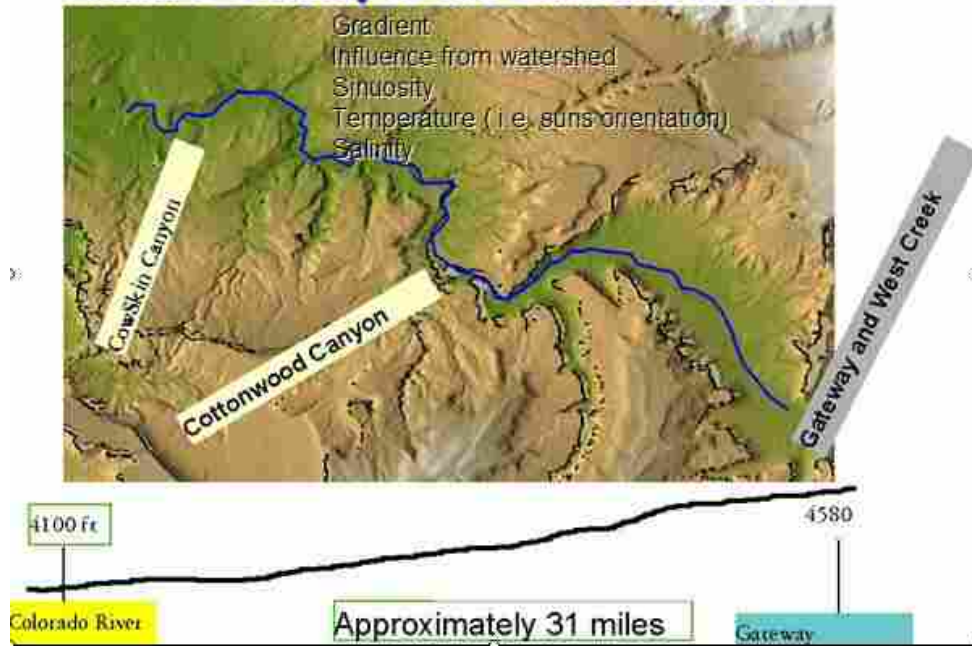
Reach 6 location

Reach 7 Saucer Basin and Gateway



Reach 7 location

Reach 8 Gateway to The Colorado River



Reach 8 location

Appendix C

Historic Flow Recommendations for the Dolores River

	Base Flows	Peak/Spawning Flows	Source
Coldwater/Trout	20cfs(dry) 50cfs (normal) 78 cfs (wet)		Dolores project EIS 1981
	50 cfs	125cfs(rainbow-spring) 65 cfs (brown-fall/winter)	BLM 1990
	80cfs (summer) 40 cfs (winter)	100,000 af	Dolores River Biology Team 1993
Warmwater/Native	80 cfs (summer) 30 cfs (winter) 200 cfs (riffle)		
	50 cfs (dry) 78 cfs (wet/normal)	Simulate natural hydrograph	Bio/West 1992
Riparian/Channel		2,000 cfs/7-days/1-in-10 yrs *500 cfs ramping	BLM 1990
		1,200 cfs	Anderson 2003
Boating	Canoe – 125 cfs Float/Fishing – 300 cfs Scenic Boating – 800 cfs Minimum Whitewater – 1,100 cfs Optimum Whitewater – 2,000 cfs		BLM 1990

Figure 16 (Historic table of flow options)

Peak Flows: Peak flows drive critical processes in natural river systems. Peak flows drive channel processes, including reshaping sediment deposits, building and maintaining floodplains, flushing fine sediments, scouring vegetation, and maintaining natural channel meanders. They also support key biological processes, including maintaining spawning habitat, providing access to in-channel, warmwater and backwater habitats, maintaining temperature, chemical, and nutrient characteristics, triggering spawning and other life-cycle stages, and regenerating riparian vegetation communities. It is critical to identify the appropriate magnitude, timing, duration, ramping, and recurrence of peak flows to support the river system.

Base Flows: Base flows sustain appropriate, year-round, in-channel habitat for fish, and maintain water levels to support riparian vegetation in the late summer period.

Appendix D Planning for Drought

Colorado Water Conservation Board May 2000

Draft

“Colorado’s climate is relatively dry and extremely variable. Annual precipitation averages only 17 inches statewide. It varies from a high of 55 inches in a few isolated high-mountain locations to a low of 6 inches in some valleys. Compared to the rest of the United States, the level of precipitation in Colorado is meager. A study of historical dry and wet periods in Colorado by the Colorado Climate Center at CSU identified five significantly severe statewide droughts from the instrumental record over the last century. The last significant multi-year drought in Colorado ended in 1978. The last two decades have seen the most reliable precipitation since before the "dust bowl" of the 1930's, but a severe, sustained drought will occur again. It is only a question of how often, how dry, and how long. less than the people, animals and plants in a region are accustomed to enjoy. As Colorado continues to grow, and as we approach the “fully appropriated”

San Miguel-Dolores-San Juan River Basins Sufficient precipitation records are available from 1898. Recent droughts in the San Miguel-Dolores- San Juan River Basins occurred in 1899-1905 (extreme), 1918-1919, 1931-1936, 1945-1947, 1951-1952, 1954-1957, 1976-1978, and 1989-1991. The most sustained drought in the tree-ring record for the San Miguel-Dolores-San Juan River Basins occurred in 1273-1289. Many archaeologists believe this period of severe sustained drought coincided with the abandonment of Mesa Verde by the Anasazi. Several shorter, but more severe droughts, were identified between AD 512 and 1673. Tree-ring Record for Southeastern Colorado(Point60),1700-1979”

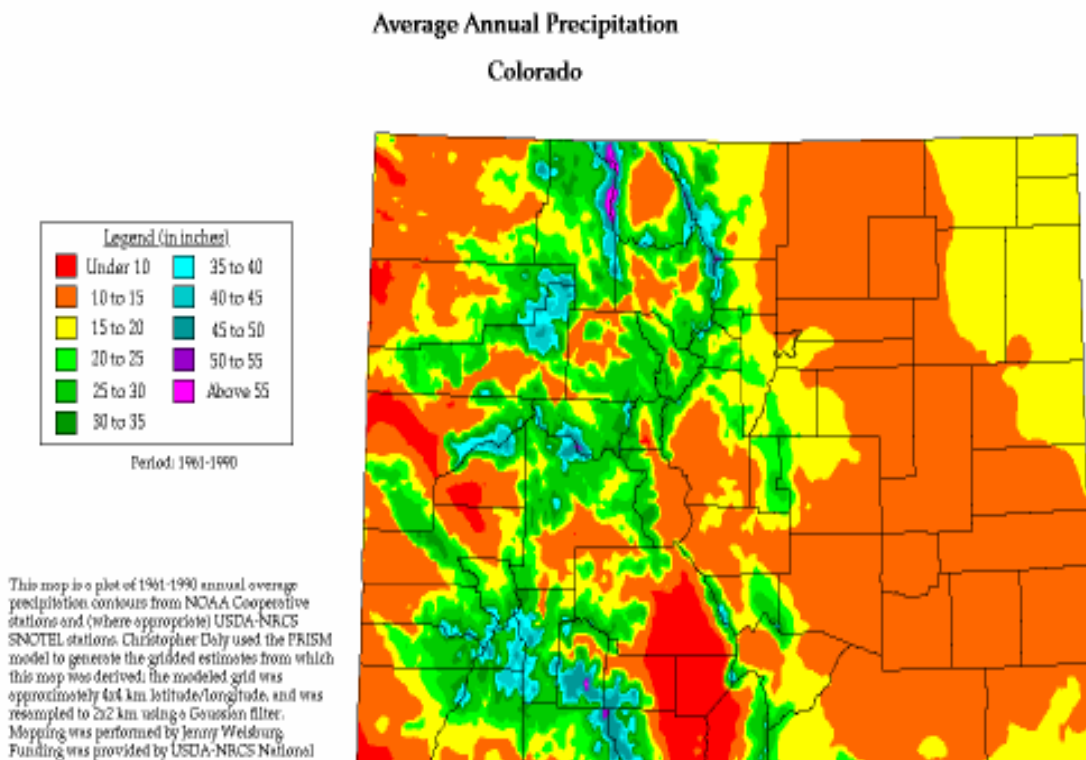


Figure 5 Map showing general precipitation for the State of Colorado

Taken from the *Draft*

Planning for Drought

Colorado Water Conservation Board May 2000

Appendix E

Vegetation Changes in a Riparian Community Along the Dolores River Downstream from McPhee Reservoir in Southwestern Colorado

David Kriegshauser and Preston Somers
Department of Biology
Fort Lewis College
1000 Rim Drive
Durango, CO 81301

Cottonwoods (*Populus spp.*) are well adapted to the fluvial dynamics created by annual spring runoff. Research indicates that river damming and controlled water flows lead to downstream riparian forest decline (Rood and Heinze-Milne, 1988; Fenner et al., 1985; Johnson, 1992; Simonin, 2001). Controlled flow of the Dolores River in southwestern Colorado began in 1984 when McPhee Reservoir was completed. In 1988 we established a study area that encompasses a riparian community dominated by narrowleaf cottonwood (*Populus angustifolia*) along the Dolores River downstream from McPhee Reservoir. Line-intercept transect and quadrat sampling methods were employed, in conjunction with repeat photography, over a 14-year period to document vegetative changes. At this point in the long-term study, the most significant change is the establishment of willows (*Salix spp.*) on the beach. Degradation of riparian communities may lead to habitat loss and disturbance in wildlife distribution (Hoover and Wills, 1987). The results of this study could possibly assist resource managers in conservation of riparian ecosystems.

INTRODUCTION

Moisture is one of the premiere limiting factors for plants and animals, and where moisture is not limiting--in streams, marshes, lakes, and the land areas adjacent to these features (the riparian zone)--one finds very distinctive communities of water dependent and drought-intolerant species. Major streams in the foothills and basins of the San Juan region commonly are lined with riparian woodlands dominated by cottonwoods (*Populus spp.*). These woodlands and associated phreatophytic vegetation (composed of plants that require a perennial water source near the soil surface) have changed dramatically during the latter half of the 19th and the 20th centuries on the western Great Plains and in the southwestern United States. Much of the change results from the fact that riparian areas are the ones that were first settled and developed for city and town sites, farmlands, and residential areas. Vast areas of riparian communities have been cleared or modified beyond recognition. Recently we see that there are significant but more subtle changes in the remaining riparian communities that were once thought to be "wild" or "natural". Here the most common pattern seen is a decline of the native cottonwoods, and their replacement by native riparian shrub species such as sandbar willow (*Salix exigua*) or exotic phreatophytes such as tamarisk or saltcedar (*Tamarix ramosissima*) and Russian olive (*Elaeagnus angustifolia*) (Miller and Bowman, 1985; Howe and Knopf, 1991).

These changes are commonly attributed to the construction of dams, water storage reservoirs, and irrigation diversion projects upstream of the affected areas. Such constructions and their

typical management regimes regulate stream flow at a relatively constant year-round level and drastically reduce spring and early summer snowmelt flows. Thus the riparian areas downstream of these water storage and management projects are no longer subjected to periodic scouring of the riverbanks and terraces. Ironically, by eliminating this natural form of disturbance and stabilizing the riparian environment, these human developments have severely impaired the reproduction of cottonwoods and have allowed shrubby phreatophytes to invade sites adjacent to the river channel that previously were bare or supported cottonwood seedlings (Johnson, 1992; Fenner et al. 1985).

Unregulated rivers typically meander in their floodplains. In the western United States these unregulated rivers carry enough water in spring floods to cause their meanders to move sideways and downstream. The current is strongest on the outside and downstream portion of the bend of a meander, called a “cut bank”, and the greatest erosion occurs here (Leopold et al., 1964). This erosion undermines and carries away alluvium that was deposited during a previous and higher meander cycle. Often there is mature cottonwood woodland on this older alluvium, and large trees are felled and swept away (Johnson, 1992). In contrast, the current is weakest on the inside and downstream portion of the bend of the meander, and as a result a point bar of alluvium is deposited there (Leopold et al., 1964). Because erosion and deposition are greater in the downstream portion of each meander, the entire meander system moves down the valley. At any given point along the floodplain there is a cycle of erosion of older deposits followed by deposition of newer deposits. This cycle is repeated over and over.

The life cycle of the cottonwood in the West fits this geomorphic process well. Cottonwoods produce large quantities of small seeds, each surrounded by light fluff. These seeds are released and easily dispersed by wind or water in early summer -- exactly the time that receding spring snowmelt floodwaters are exposing newly created point bars and beaches (Fenner et al. 1984). Seeds landing on these moist sites germinate promptly and begin to grow. Cottonwood seeds have a limited period (1 to 5 weeks) of viability after their release, but once they germinate and are established as seedlings their roots may grow an average of 0.25 inch (6 mm) per day (Fenner et al. 1984). This rapid growth rate allows the roots to follow and maintain contact with the receding water table during the summer, and to reach a depth of 29-65 inches (72-162 cm) by the end of the growing season (Fenner et al., 1984).

Cottonwoods also can reproduce vegetatively in their early years. Once established, cottonwoods grow quickly (6-13 ft (2-4 m) annually under ideal conditions), and stands thin themselves naturally. The trees achieve their maximum diameter and height in 40-50 years (Hoover and Wills, 1987). During this time the water table level under a typical cottonwood stand is lowered through degradation of the river channel by erosion. Accumulation of silt on the ground surface may also lift the substrate above the water level. As the meander system moves down the floodplain, the distance between the cottonwood stand and the bend that laid down its substrate increases. At the same time, the distance between the stand and the approaching upstream bend decreases. Eventually the upstream bend encroaches on the stand, erodes away its substrate, and carries away the trees. By this time the stand typically has reached maturity or decadence (Johnson, 1992).

In the regulated state that most rivers in the western United States are now found, this natural geomorphic process is halted or greatly retarded (Rood and Heinze-Miline, 1988). Under the regime of constant flow and low volume discharge, erosion of older deposits and deposition of new alluvium occur at a negligible rate. Scouring of banks does not occur. Water coming from reservoirs is low in silt and sand content; thus siltation does not occur (Simons, 1979). Indeed, silt may be removed from the river channel by the clear water from reservoirs in reaches immediately below these impoundments, and this process results in "armoring" of the river channel by leaving a bed lined with cobbles and pebbles (Simons, 1979).

Thus there is stability in a system that once was characterized by dynamic movements of alluvium. Cottonwoods are well adapted to take advantage of the dynamics of fluvial processes. They are not favored by stability. Their roots need to be in contact with the water table, but the water table falls and stays low in the floodplains of many regulated streams. The life span of cottonwoods is short, and they do not reproduce well vegetatively in their mature stages. There are no new point bars and beaches where seedlings may become established in managed stream systems. Cottonwoods are intolerant of shading and competition by other plants (Hoover and Wills, 1987). Shrubby phreatophytes invade stream banks in the absence of bank scouring, and eliminate the opportunity for cottonwoods to gain a foothold on the areas closest to the water table. We find today that cottonwood woodlands of the high plains and of much of the Colorado River drainage are maturing and going into decadence with poor recruitment of younger individuals. Often lining the banks of these rivers are exotic phreatophytes (e.g., saltcedar) that support a lower diversity of native fauna and favor introduced animal species (Cohan et al., 1978).

We can see an example of this process in the Dolores River, which has its headwaters in the Lizard Head Wilderness of southwestern Colorado. This river flows south for 67.5 miles (108.6 Km) to the town of Dolores where it turns northward. In 1984 the McPhee Dam was completed 17 miles (27.4 Km) downstream of Dolores, and filling of McPhee Reservoir began. This on-stream reservoir was designed and built as the primary water storage facility of the Dolores Project by the United States Bureau of Reclamation. It supplies water to this large regional irrigation system as well as municipal and industrial water for several local towns. Volume discharge from McPhee Reservoir is now regulated at three levels: 25, 50, and 75 cubic feet per second (CFS) (0.7, 1.4, and 2.1 m³/s) (Hoover and Wills, 1987). During the spring there are brief larger releases of 1,200 CFS (33.7 m³/s) or more that allow for white water recreation. In contrast, the Dolores River at Dolores had sustained peak flows of up to 8,000 CFS (224.1 m³/s) with a mean peak of 3,360 CFS (94.4 m³/s) in the ten years before the filling of McPhee Reservoir. The mean annual flow at the town of Dolores was 438 CFS (12.3 m³/s) with a historic minimum of 8.0 CFS (0.2 m³/s), but because of the Montezuma Valley Irrigation system diversion just downstream of Dolores, the river was often dry downstream of this diversion during the late summer (Hoover and Wills, 1987). The flow regime in the Dolores River downstream of McPhee Dam is now practically constant and quite low compared to levels during most months before construction of the dam.

The valley downstream of McPhee Reservoir contains much riparian woodland that is heterogenous in age class and diverse in floral and faunal composition. Narrowleaf cottonwood (*Populus angustifolia*) is the dominant tree species, and stands of any age within its typical

lifespan of 80 to 90 years may be found. These stands with their associated trees and shrubs have been developing and interacting with the fluvial processes of an unregulated stream for many years. However, there is concern that these narrowleaf cottonwood woodlands will decline as they naturally age without recruitment of new individuals into the population. The present and future stable, low flows of the Dolores River downstream of McPhee Reservoir will likely eliminate the fluvial dynamics upon which cottonwoods are dependent, and allow the invasion and establishment of shrubby phreatophytic communities along the river.

METHODS

A single study area was chosen that is representative of the riparian woodlands downstream of McPhee Dam. We gave this area the name "Lone Dome Study Area" because of its proximity to a prominent rock outcrop of that name that can be clearly seen just upstream. This study area covers approximately 27 acres (11 ha) at an altitude of 6,520 - 6,560 feet (1,987 - 1,999 m). It is located in Montezuma County, Colorado, in the NE 1/4 of S23 and NW 1/4 of S24, T39N, R17W. The Dolores River meanders to a limited extent within its floodplain in this portion of the canyon which is about 700 feet (213 m) deep here and is itself composed of gentle entrenched meanders. The study area is enclosed by an oxbow bend and is naturally bounded on three sides by the river channel and on the fourth or south side by the steep canyon wall. This area is across the river from the public road that follows this portion of the canyon bottom.

On July 11 - 13, 1988, one of us (Somers) conducted a preliminary study of this area with students as a class project. Using meter tapes we conducted 18 line intercept transects each of which was 30 m long and arranged end to end so that the entire set of transects was arranged in a straight line 540 m long. This line began at the edge of the river and proceeded along a compass bearing of S60E to its end at the bank of the river around the bend upstream. Thus the line of transects started in an area of fresh deposition of alluvium and proceeded through successively older communities to an area of erosion of a cut bank. The beginning and end of each transect was permanently marked by driving a 1/2 inch (13 mm) diameter by 1 m long steel reinforcing bar into the ground. Each bar was identified with an aluminum tag that gave the beginning and ending position of that transect along the 540 m line. Each shrub and tree that intersected the line was recorded. In 1993 we established 16, 1 m² circular quadrats along an extended point bar beach. Each quadrat was centrally marked with steel reinforcing bar. Every shrub and tree within each quadrat was recorded and percent cover was estimated via team consensus. Photographs were taken of the study area in late morning from three prominent rock outcrops on the canyon wall opposite the study area. Data were recollected every July from 1988 through 2001, except for the year 1992. Regression analysis was applied to this data using SPSS statistical software. We used enlarged black and white aerial photographs of the study area as templates to translate 3" x 5" false color infrared aerial photographs into vegetation maps. The photographs were taken in 1983 (pre-dam) and 1996 (12 years post-dam). A transparent grid with 100 squares per square inch was placed over the vegetation maps, and we determined the area by counting the number of squares occupied by each vegetation type. We then calculated the amount of change in area shown by each vegetative community from 1983 to 1996. Finally, volume discharge records for the Dolores River at Dolores and Bedrock were downloaded from the USGS website and imported into Microsoft Excel.

RESULTS

Analysis of vegetation maps constructed from infrared aerial photographs of the study site from 1983 and from 1996 revealed that bare beach and sandbar decreased by 95%; shrubby vegetation on the beaches, point bars, and up to 60 m inland increased by 665%; young cottonwood groves near the beaches decreased by 90%; meadows in 1983 (indicated by pink tint) were xeric meadows (indicated by blue tint) in 1996.

Peak volume discharge of the Dolores River due to spring snowmelt in April – June has been reduced by at least 36.4% in Bedrock, Colorado (see Figure 1). Percent cover of willows (*Salix spp.*; primarily *S. exigua* with occasional *S. amygdaloides*) has increased significantly in the 16 beach quadrats (Figure 2; $P=0.000$; $R^2=0.932$). Percent cover of cottonwood (*Populus angustifolia*) has not increased significantly in the 16 beach quadrats (Figure 2; $P=0.807$; $R^2=0.0091$). Number of individuals of willows has increased significantly in the first 30 m of the line intercept transect (Figure 3; $P=0.001$; $R^2=0.661$). Number of individuals of cottonwood has not increased significantly in the first 30 meters of the line intercept transect (Figure 3; $P=0.404$; $R^2=0.0041$). Silverberry (*Shepherdia argentea*) was observed from 1989 through 1997 on a portion of the line intercept transect adjacent to an old flood-carved channel. No observations of silverberry have been documented from 1998 through 2001 on this transect.

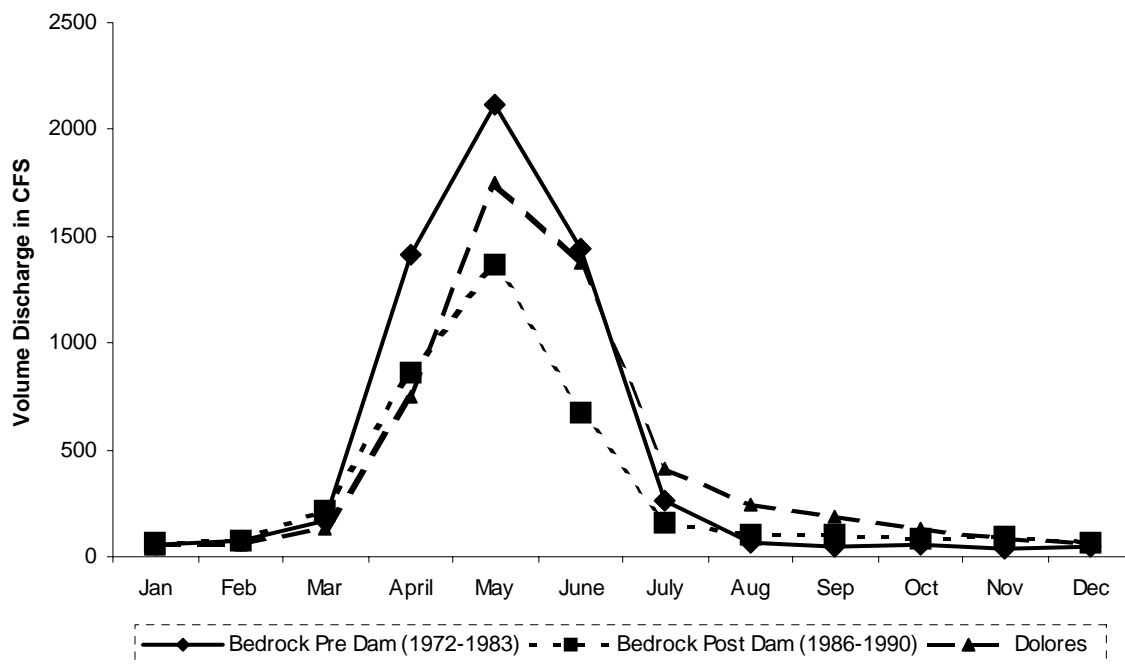


Figure 1. The effect of regulated flow is illustrated by the annual volume discharge of the Dolores River at Dolores (flowing into McPhee Reservoir) and at Bedrock (approximately 60 miles downstream from McPhee Dam). Volume discharge at Dolores is a 100 year average.

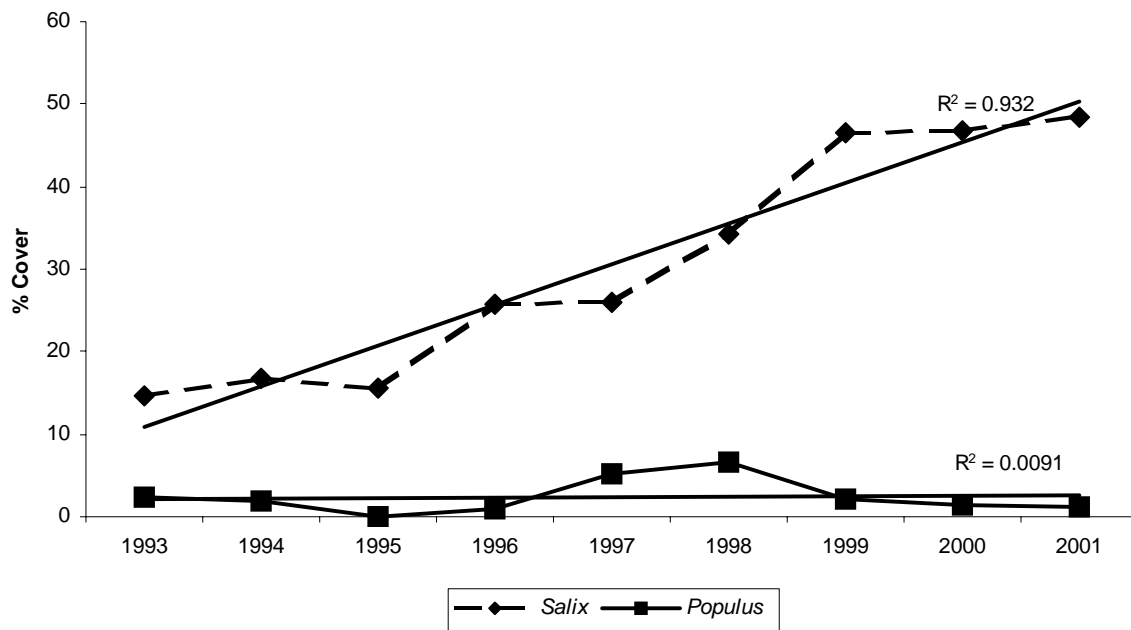


Figure 2. The mean % cover of cottonwood and willows on the beach as indicated by 16 circular 1m² quadrats along an extended portion of beach.

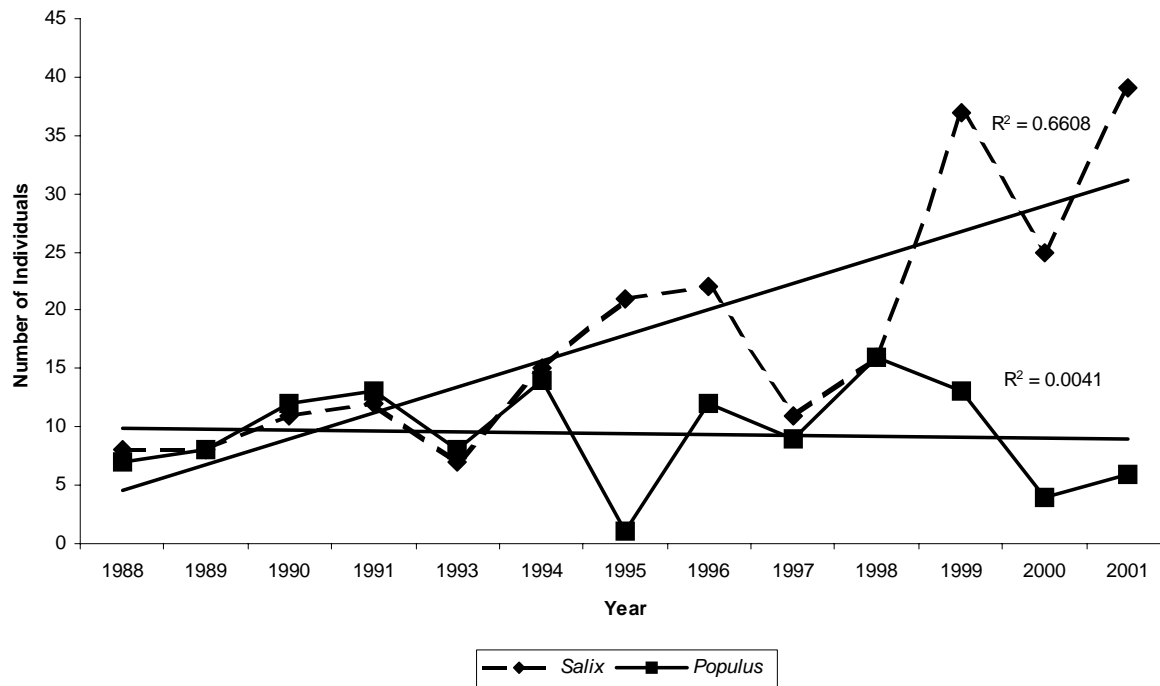


Figure 3. The number of individuals of cottonwood and willows in the first 30 m of the line intercept transect.

DISCUSSION

Rood and Heinze-Miline (1988) used aerial photographs to quantify riparian forest decline along three parallel rivers in Alberta, two of which were dammed. The first river had been dammed for 20 years and showed a 48% reduction in riparian forest. The second river had been dammed for 17 years and showed a 24% reduction in riparian forest. The third undammed river showed less than 1 % riparian forest reduction over a 20 year period. Furthermore, all three rivers showed only slight decline in riparian forest above the dams. The locations of these study sites and the extent of forest decline supports a causal relationship between river damming and forest reduction (Rood and Heinze-Miline, 1988).

Figure 1 illustrates that regulated flow of the Dolores River has greatly reduced volume discharge in Bedrock—approximately 60 miles downstream from the dam. Over a nine year period the percent of beach covered by narrowleaf cottonwood has shown no significant change in our study area while the percent of beach covered by willows has increased significantly (see Figure 2). This indicates that although some cottonwood seedlings begin to grow each year within the beach quadrats, they are failing to flourish. Cottonwood seedling establishment is optimal upon a freshly deposited point bar or scoured beach (Fenner et al. 1984; Johnson, 1992) for they are intolerant of shading and competition (Hoover and Wills, 1987). It is to be expected that a sharp increase in the percent cover contributed by willows will inhibit the establishment of cottonwood seedlings on the beach.

Over a thirteen year period the number of individuals of narrowleaf cottonwood has shown no significant change in the first 30 meters (the beach portion) of the line intercept transect. This steady state indicates that there is neither seedling establishment nor vegetative propagation in any measurable amount. Willows, on the other hand, have increased their numbers along the beach transect, but only in recent years. In this case the failure of cottonwoods to recruit new individuals into the population cannot be attributed to competition from willows. Figure 3 indicates that cottonwoods were not flourishing even before willows became dominant. We suggest that the drastic reduction in the magnitude and duration of the spring flood may have lowered the water table to a level too low for adequate access by the young cottonwoods within 30 m of the beach. Draw down of ground water at critical growth periods may stunt or kill young cottonwoods by simulating conditions of extended drought (Johnson, 1994; Fenner et al. 1985). Greatly retarded fluvial dynamics and alluvial deposition, along with a lowered water table will lead to depletion of available nutrients (Fenner et al. 1985; Simonin, 2001). These changes will also lead to increased mortality among mature cottonwoods and reduction in cloning opportunities (Johnson, 1994) further inland from the river channel.

Colonization of point bar and open beach by willows is clearly revealed in repeat photography (see Figures 4 and 5). The pre-dam and post-dam infrared aerial photographs also indicate that inland meadows may be changing from mesic to xeric conditions. Reduction in spring flood intensity prevents deep inundation of the flood plain (Fenner et al. 1985). Pre-dam peak flow of up to 8,000 CFS would have inundated these meadows, which are only 3 to 6 vertical feet above the water level at 75 CFS. Regulated spring discharge from McPhee Reservoir averages 1,200 CFS or less for a brief period for white water recreation. This volume is probably insufficient to inundate the inland meadows. Further evidence for the drying of these meadows is revealed by

the inland portion of the line intercept transect. Silverberry (*Shepherdia argentea*) was observed from 1989 through 1997, but no observation of this species has been recorded since then.

In years preceding the construction of McPhee Dam, the Dolores River often ran dry in the Lone Dome area in the late summer due to irrigation demands when volume discharge was low (Hoover and Wills, 1987). In spite of the irrigation diversion from the river itself, spring floods still raged through the valley at near full intensity and duration. Healthy cottonwood woodland with heterogeneous age stratification existed before dam construction. Now the factors are reversed. The river flows year round, but the intensity and duration of spring flooding has been greatly reduced. Desertification of inland meadows and failure of recruitment of young cottonwoods near the beach indicate that spring floods, rather than constant low volume discharge, recharge the water table.

Colorado's cottonwood-willow riparian ecosystems support the highest wildlife species density and richness in the state (Snyder and Miller, 1992). A Division of Wildlife report by Hoover and Wills (1997) included wildlife inventories taken in the Lone Dome region of the Dolores River. They documented 24 species of amphibians and reptiles; 151 species of birds; and 34 species of mammals. Among these animals is the red bat, eastern cottontail, and wood duck, which are obligate cottonwood woodland inhabitants in Colorado (Graul and Svoboda, 1987). They will not tolerate displacement. Furthermore, fifty-one of the birds found in the cottonwood woodlands are migratory. They use these sparsely distributed riparian corridors of the Southwest to complete their annual transcontinental and intercontinental journeys. If the riparian tree component (i.e. cottonwoods) declines on a long-term, widespread basis, there likely will be a significant decline in wildlife species richness (Graul and Svoboda, 1987). The population of the Yellow Warbler, an avian species of special concern in Colorado, is declining in Southwestern Colorado. This decline is likely due to the loss of riparian habitat—for they prefer to nest in cottonwood woodlands (Ortega and Ortega, 1999). Johnson (1994) projects a shift in forest composition toward later successional species as a result of losing cottonwood colonization sites due to a decline in meandering following dam construction. Later successional species in the Lone Dome study area are hawthorn (*Crataegus rivularis*), box elder (*Acer negundo*), and eventually Gambel oak (*Quercus gambelii*). At that point, restoration would call for large-scale clearing, as cottonwood-willow communities do not successfully regenerate under such forest conditions (Johnson, 1994). Moreover, it is likely that the community that replaces the cottonwood-dominated woodland will not support the high avian diversity found there today (Hoover and Wills, 1987) because the smaller tree size of equilibrium forests provide fewer nesting cavities (Johnson, 1992). Avian species diversity has been related to diversity within foliage height (Ortega and Ortega, 1999). It is clear that preservation of cottonwood woodland is critical for the maintenance of biodiversity within and around Colorado.

Healthy ecosystems have repair mechanisms, but damage can exceed their capacity for self-restoration (Whisenant, 1999). Damaged hydrologic and vegetation regeneration processes contribute to a positive feedback system that increases riparian degradation (Whisenant, 1999). Such ecological consequences have been observed here on the Dolores River, in the Grand Canyon, in the Black Canyon of the Gunnison, in Canada, and on other continents (Ward and Stanford, 1979). It will be necessary to reintroduce a more intense spring flooding to maintain

the integrity of this dynamic ecosystem. This would be a difficult goal to achieve in the case of the Dolores River and many western rivers as this spring floodwater has been allocated for irrigation and municipal and industrial use. Also, humans have moved onto the floodplain on many dammed rivers in the West. Reintroduction of floods large enough to restore meandering could result in property devaluation or loss. In any case, simulation model results (Johnson, 1992) and evidence from our study indicates that remaining cottonwood woodlands in the Dolores River valley cannot be maintained under the current regime of low volume discharge from McPhee Dam. The solution to riparian degradation lies in the cooperative efforts of landowners, land managers, and ecologists.

ACKNOWLEDGEMENTS

The following people contributed valuable assistance in the field or encouragement in the writing process: Judy Somers, Heather Kriegshauser, Chester Anderson, Michael Highland, Barbara Swain, Dr. Albert Spencer, Dr. Catherine Ortega, Dr. Shere Byrd, and Fort Lewis College Field Ecology students from 1988 - 2001

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APPENDIX G

HABITAT REQUIREMENTS OF JUVENILE SALMONIDS

TOWARDS ECOLOGICALLY-BASED FISHERIES MANAGEMENT IN BOREAL STREAMS

Aki Mäki-Petäys

Department of Biology

Abstract

For effective management of stream salmonids, it is essential to (i) assess the productive potential of a stream in relation to species-specific habitat requirements, and to (ii) identify the key factors underlying any bottleneck periods during the life cycle of a fish. For this purpose, this PhD-thesis focuses on the mechanisms of habitat selection by juvenile salmonids in boreal streams.

Habitat preference curves for depth, water velocity, substrate and instream cover for brown trout (*Salmo trutta* L.) in river Kuusinkijoki, northeastern Finland, indicated that larger trout preferred deeper stream areas than age-0 trout did. In summer, all size-classes of trout preferred small substrates, whereas in winter, areas with cobble-boulder substrates were preferred. Winter presents a bottleneck period for trout in boreal streams; therefore winter habitat curves should be incorporated into habitat-hydraulic models when estimating habitat suitable for riverine trout in areas with severe winter conditions. The preference curves of age-0 trout were validated by correlating age-0 trout density with habitat availability at multiple sites in two boreal rivers where trout densities were monitored in 1988-1995. Substrate preference curve was effective in predicting trout densities among sites, whereas among-year variation in trout densities was best predicted by depth-related preference curves.

The responses of age-0 brown trout and grayling (*Thymallus thymallus* (L.)) to enhancement structures were investigated in artificial stream flumes. For both species a crucial habitat factor was the availability of flow refuges, especially in winter. In another experiment, age-1 trout dominated over age-0 trout when competing for velocity and overhead cover they both found suitable, emphasizing the role of intraspecific interactions in habitat selection by trout. These results suggest that the provision of a broad diversity of microhabitats should be a major goal in rehabilitation programs for fishery purposes.

A new method, combining GIS-assisted (Geographical Information System) approach with geostatistical tools, facilitated the detection of fish distribution patterns in a spatially heterogenous stream habitat. The method will likely prove valuable when determining appropriate sampling scale(s) for future studies of fish habitat selection in relation to benthic prey. Unlike Arctic bullhead (*Cottus poecilopus*, Heckel), trout did not show any aggregation

with their benthic prey according to spatially-referenced data on the distribution of lotic fishes and benthic macroinvertebrates within a stream reach.

Acknowledgements

This work was carried out at the Finnish Game and Fisheries Research Institute's Kainuu Fisheries Research and Aquaculture, Paltamo, and in the Department of Biology, University of Oulu. I wish to thank the Head of the Kainuu Fisheries Research and Aquaculture, Pekka Hyvärinen, and Prof. Markku Orell for providing me with excellent working facilities in Paltamo and Oulu, respectively. The staffs of Kuusamo Aquaculture and Oulanka Biological Station are acknowledged for their logistical help. This work is a contribution from the Finnish National Graduate School on Fish Biology and Fisheries. It was also supported by grants from the Rural Business District of Kainuu Fisheries Section and the Academy of Finland (project no. 33430 to Prof. J. Hooli).

I wish to express my deepest gratitude to Doc. Timo Muotka as being the supervisor of my thesis: in addition to his contributions to the all papers, to adopt some of his scientific enthusiasm and knowledge in stream ecology will certainly be a cornerstone for my research work also in the future. The intellectual influence of Dr. Ari Huusko was essential in my learning process on the ecology of stream salmonids. Doc. Kalevi Kuusela originally guided me to the fascinating world of streams and their living organisms.

I owe my special thanks to my collaborators, Dr. Teppo Vehanen and Petri Kreivi. Long-lasting and inspiring discussions with them in the field and the laboratory have been fruitful and constructive at many phases of my studies. I also appreciate the aid and encouragement by Dr. Pertti Tikkanen in designing and analyzing the laboratory experiments of this thesis. Risto Kannel and Arto Juntunen provided logistical help with laboratory studies, and Pekka Korhonen and Olli van der Meer assisted with field work. Members of our Stream ecology group, Pia Hentunen, Riku Paavola, Antti Juntunen, Arto Huhta, Pekka Laasonen, Jani Heino and Kristian Meissner provided stimulating and exciting, yet unpredictable, atmosphere during the extensive field work periods at the Oulanka Biological Station.

I am deeply grateful to my parents; they understood and supported me throughout this work. Finally, I want to thank my sons Ville and Miika, and also my ex-wife Mari, for their patience. Living and growing together with my sons balanced my life between the scientific world and everyday cares, and kept me going during many long evenings with this thesis.

This thesis is based on the following papers, which are referred to in the text by their Roman numerals:

- I Mäki-Petäys A, Muotka T, Huusko A, Tikkanen P & Kreivi P (1997) Seasonal changes in habitat use and preferences by juvenile brown trout, *Salmo trutta*, in a northern boreal river. Can J Fish Aquat Sci 54: 520-530.

- II Mäki-Petäys A, Muotka T & Huusko A. Densities of juvenile brown trout, *Salmo trutta*, in two subarctic rivers: assessing the predictive capability of habitat preference indices. *Can J Fish Aquat Sci*, in press.
- III Mäki-Petäys A, Vehanen T & Muotka T. Microhabitat use by age-0 brown trout and grayling: seasonal responses to streambed manipulation under fluctuating flow. Manuscript, submitted.
- IV Vehanen T, Mäki-Petäys A, Aspi J & Muotka T. Intercohort competition causes spatial segregation in brown trout in artificial streams. *J Fish Biol*, in press.
- V Muotka T, Mäki-Petäys A, Kreivi P & Högmander H (1998) Spatial associations between lotic fish, macroinvertebrates prey and the stream habitat: a multi-scale approach. *Boreal Env Res* 3: 371-380.

Chapter 1. Introduction

During the last two decades it has become obvious that the study of running waters cannot be carried out by excluding the role of human activities, nor is it possible to ignore the urgent need for conservation (Allan 1995). At the same time, stream ecology has taken long steps from a descriptive to a more theoretically-oriented science (Muotka 1994). Describing repeated patterns of stream organisms, populations and communities in relation to their biotic and abiotic environments is a prerequisite for testing general ecological theories in stream communities. This may in turn provide mechanistic explanations for anthropogenic changes in stream and river ecosystems. Such information is especially important, if the changes are general and controlled by few key factors, and if these factors can be affected by management (e.g. Fausch *et al.* 1988). Therefore it is evident that we need to increase our knowledge about the potentially complex habitat requirements of stream-dwelling organisms to facilitate mitigating the negative effects of human impacts on the stream environment.

Many salmonid fish species are born in running waters and, after smoltification, they migrate to a lake or sea to grow, returning to their natal river to spawn. Some salmonids or stocks of them, however, reside in streams all of their life (Wootton 1990). In contrast to the classical view of salmonid parr as site-specific stream dwellers (for reviews, see Gowan *et al.* 1994, Fausch & Young 1995), recent observations of stream salmonids have also documented extensive habitat shifts from their nursery areas to varying parts of a watercourse (estuaries, tributaries, lakes), depending on environmental conditions (e.g. Erkinaro 1997). Unfortunately, damming rivers for hydropower purposes and dredging them for log floating, have affected the juvenile lifestages of stream salmonids negatively. Management plans to restore impacted rivers should provide spawning environments, as well as suitable rearing habitats for juvenile salmonids to re-establish the threatened or extinct salmonid populations in streams (Heggenes 1989; see also Fausch & Young 1995, Erkinaro & Gibson 1997).

Many fish enhancement projects have focused on improving the stream habitat for salmonids by increasing the streambed complexity, and thereby providing more instream cover for salmonids (e.g. Moore & Gregory 1988, Binns 1994, Gowan & Fausch 1996, Sundbaum & Näslund 1998). Extensive projects have also been launched in Finland to restore rivers formerly dredged and

channelized for log transport. The goal of these projects is to create a more heterogeneous stream habitat, aiming at the well-being of salmonid fishes and their food organisms (Laasonen *et al.* 1998, Yrjänä 1998). Habitat-hydraulic models are commonly used to estimate the quantity of habitat suitable for salmonids at different stream flows (e.g. Heggenes 1996), and they may also be used as an evaluation tool for different enhancement projects (e.g. Shuler & Nehring 1994). This thesis is a part of an ongoing project established to assess, by the means of habitat-hydraulic modeling, how successful these rehabilitation programs have been in satisfying the habitat requirements of juvenile salmonids (see Huusko & Yrjänä 1997).

Stream salmonids undergo marked seasonal (e.g. Cunjak & Power 1986, Heggenes *et al.* 1993) and ontogenetic (Bohlin 1977, Moyle & Baltz 1985, Greenberg *et al.* 1996) habitat shifts; therefore the season- and size (age)- specific knowledge of their habitat requirements is essential for adequate modeling. To study these questions both field and laboratory approaches were adopted in this thesis. First, I developed season-specific (summer, autumn, winter) preference curves for depth, mean water velocity, substrate size and cover of aquatic vegetation for three size-classes of juvenile brown trout (*Salmo trutta* L.) in a northern Finnish river (I) to facilitate regionally relevant application of habitat-hydraulic models in Finland (see Huusko & Yrjänä 1997). Then I evaluated the accuracy of these preference curves in predicting the densities of age-0 trout in two subarctic rivers (II). In artificial stream channels, I examined the microscale behavioural responses of age-0 brown trout and grayling (*Thymallus thymallus* (L.)) to streambed complexity under varying flows (III).

Habitat-hydraulic models have at times met with limited success, probably because spatially and temporally varying biological factors have not been incorporated into the models (e.g. Orth 1987), and because the models may operate at a larger spatial scale than that relevant to fish (e.g. Heggenes 1996). It is obvious that more information about these issues with respect to the habitat selection of fish, is required for biologically more realistic habitat models. Accordingly, the purpose of paper IV was to study whether intercohort competition may cause spatial segregation among size-classes of juvenile trout, and whether this segregation is seasonally variable. Finally, because the appropriate scale for a study is often difficult to determine *a priori* (e.g. Muotka & Penttinen 1994), a multi-scale approach was adopted (V) to examine the spatial association between juvenile brown trout, Arctic bullhead (*Cottus poecilopus*, Heckel), macroinvertebrate prey and the stream habitat.

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• 2.1. Study area

The field work was conducted in rivers Kuusinkijoki (I, II and V) and Kitkajoki (II). These rivers are 3rd-order tributaries to the river Oulankajoki, near the Arctic Circle in northeastern Finland (a detailed map appears in paper II). Both rivers flow through spruce and pine forests and discharge into Lake Paanajärvi, and finally to the White Sea in Russia. The water quality of river Kuusinkijoki is slightly degraded by human activities, whereas river Kitkajoki is in a near pristine condition (Koutaniemi & Kuusela 1993). Water temperature peaks at about 17-19°C in late July to early August and falls below 10°C in late October. Both rivers are mainly ice-covered from mid-November to late April. The hydrology of the study rivers is characterized by a high spring flood in May and a smaller flood in late September - early October. Compared to the flow regime of River Kuusinkijoki (mean: 10 m³/s, range: 1-60 m³/s), River Kitkajoki is less variable (mean: 21 m³/s, range: 8-50 m³/s). Brown trout and Arctic bullhead are the dominant fish species in both rivers, although other species, especially European grayling and European minnow (*Phoxinus phoxinus* (L.)), also occur in these rivers.

• 2.2. Laboratory streams

The experimental studies were performed using hatchery reared salmonids in three (IV) or four (III) identical indoor flumes located at the Finnish Game and Fisheries Research Institute, Kainuu Fisheries Research and Aquaculture, northern Finland. The flumes were 6 m long and 37 cm wide, with a maximum depth of 30 cm. Water from the hatchery main storage was transferred by two submersible pumps into two large containers (2000 l each), and both containers supplied water into two of the flumes. Fish were always allowed to acclimate 48 h before they were used in the trials, to allow them to recover from the handling stress and transportation. Artificial light was provided on a 12:12 light:dark photoperiod (room light, 125 lx: red light 3 lx). Three (IV) or four (III) video cameras per flume connected to video monitors facilitated observation of fish from an adjacent room. Observations were made either only in daytime (III), or during both day and night (IV). Both experiments were conducted in summer and winter.

2.3. Seasonal and ontogenetic changes in habitat preference of trout (I)

Juvenile brown trout were located by means of underwater observation or modified point electrofishing procedure (e.g. Moyle & Baltz 1985) and divided into three size-classes (4-9, 10-15, 16-22 cm). Habitat characteristics were measured from each fish location in six separate stream sections (range of areas: 600-3000 m²), and to quantify habitat availability, the same variables were measured from the same sampling sections using a stratified random sampling protocol. Season-specific preference curves were developed by relating use to habitat availability for four habitat variables: depth, mean water velocity, dominant substrate size and cover of aquatic vegetation. A two-way ANOVA was conducted to test for potential season x size-class interaction effects in trout habitat use. In addition, discriminant function analysis (DFA) was employed to examine the multivariate nature of seasonal variation in habitat availability and habitat use by trout. Principal component analysis (PCA) was used to examine whether habitat use differed between trout size-classes, and whether trout habitats constituted a nonrandom subset of the habitat available in the study reach.

2.4. Predictive capability of habitat preference indices (II)

Fourteen to 25 stream sites (mean area \pm 1 SE: 97 ± 3.1 m²) were electrofished during eight summers using the three-pass removal method, and fish densities were calculated according to Bohlin *et al.* (1989) to obtain estimates of late summer densities of age-0 brown trout. Habitat characteristics (depth [d];, dominant substrate size [s];, surface water velocity [v];) were quantified at each site immediately after electrofishing. Trout densities at each study site were correlated with site-specific suitability values, ranging from 0.0 (unsuitable) to 1.0 (optimal). These suitability values were obtained by converting the mean value of the measured habitat variables at each sampling site to preference indices based on the corresponding preference curves for age-0 trout in summer (P[d];, P[v];, P[s];). Densities were also correlated with calculated composite preference indices P[dv];, P[ds];, P[dsv]; (see Gan & McMahon 1990). To assess the importance of substrate for trout during their first year of life, a ratio called ‘apparent survival’ (the density of age-1 trout at a site in year *t* vs the density of age-0 trout at the same site in year *t*-1, see Gowan & Fausch 1996) was correlated with summer and winter indices of substrate preference for age-0 trout and with the same index for age-1 trout in summer.

2.5. Microscale behavioral responses of salmonids to stream enhancement structures (III)

Microhabitat selection by age-0 brown trout and grayling was investigated in experimental flumes with either ‘channelized’ or ‘restored’ streambed structures. Fish were exposed to low- and high-flow treatments in both summer and winter. Water velocities used by fish, Donnelly’s (1978) index of aggregation, and the distances from each fish position to the (i) nearest stone and (ii) the inlet of the flume, represented dependent variables in the study. The experimental design incorporated 8 treatment combinations in a 2 x 2 x 2 factorial structure. Because the same fish were observed during both low- and high-flow treatments, flow was a within-subject factor in repeated measures ANOVA, whereas season and flume type represented between-subject factors. Species- and season-specific preference curves for bottom water velocities were developed in both flume types by relating velocities used by fish to those available in the flumes.

2.6. Intercohort competition of juvenile brown trout (IV)

The role of intercohort competition in causing spatial segregation between age-0 and age-1 brown trout was examined in artificial flumes during two seasons (winter vs summer) and at different times of day (day vs night). Competition treatments were: (1) two small trout, (2) two small and two large trout, and (3) two large trout per stream channel. The channels were longitudinally divided into three experimental units (1 m each), each of which consisted of two velocity shelters on the opposite sides of the stream, one of the shelters providing overhead cover for the fish. Time of day was a within-subjects factor, whereas season and treatment were between-subject factors, when differences in swimming movements, aggressions and velocity use by trout were analysed using repeated measures ANOVA. Differences in shelter use by trout size-classes were tested by logit analysis (Christensen 1990), where the proportion of a habitat type (velocity shelter, shelter with cover, open stream) used was the response variable, and treatment, season, time of day and fish size (small vs large) were the explanatory variables.

2.7. Spatial scales of lotic fish and their habitat (V)

A multiscale approach was adopted to study the spatial association between stream fish, their food resources and habitat characteristics. The sampling area of 23 m x 4.5 m was divided into 279 grid cells, each 0.75 m x 0.5 m. The abundance of brown trout and Arctic bullhead were quantified within each cell by the means of point electrofishing (see paper I), and habitat variables (depth, water velocity, substrate size and instream vegetation) were also measured from each cell. Benthic samples were collected from paving bricks (14 cm x 13 cm) in the centre of every other cell. The spatial patterns of habitat variables and invertebrate abundances were visualized by colored contour plots, and fish distributions were superimposed on these plots. Two geostatistical methods, semivariogram and kriging (see Rossi *et al.* 1992), were also used to study the spatial patterns among the data.

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3.1. Habitat requirements of juvenile trout (I)

Larger trout generally preferred deeper stream areas than young-of-the-year fish (I, see also paper V). When observed over seasons, the optimal ranges for depth were 5-35, 40-60, and 50-75 cm for 4-9, 10-15, and 16-22 cm trout, respectively (I). This bigger fish - deeper habitat pattern has been documented in numerous studies of habitat selection by stream fish (e.g. Bohlin 1977, Power 1987, Harwey & Stewart 1991). In late summer and autumn age-0 trout favored stream areas with large amounts of aquatic vegetation. The largest trout (16-22 cm) occupied habitats with little vegetation cover throughout the year, and in winter, all trout avoided areas with a high percentage of aquatic vegetation. At the onset of winter, all trout size-classes moved into shallower water, but this mainly reflected seasonal variation in habitat availability. Cunjak and Power (1987) and Heggenes *et al.* (1993) suggested that instead of depth, velocity refuges and overhead cover are probably the primary factors in winter habitat selection of trout. In winter, trout preferred slowly flowing stream areas, whereas in other seasons the mean water velocities used by trout paralleled habitat availability. In summer, all size-classes of trout preferred small substrate sizes, whereas in winter, areas with cobble-boulder substrate sizes were preferred, especially by trout larger than 10 cm. Due to the reduced swimming ability of fish at low water temperatures (e.g. Rimmer *et al.* 1985, Graham *et al.* 1996), the interstitial spaces of coarse substrates may be the prime determinant of the suitability of a stream area as wintering grounds for salmonid fish (Rimmer *et al.* 1984, Heggenes *et al.* 1993).

Discriminant analysis ranked water velocity and depth as the most important variables in differentiating among habitats used by trout at different seasons. Similar rankings of physical habitat variables in trout microhabitat use have been reported by Gatz *et al.* (1987) and Heggenes and Saltveit (1990). When observed across seasons, principal component analysis revealed the

clearest pattern of nonrandom habitat use for the smallest size-class of trout. The microniches of these fish were characterized by slower water velocities, lower depths, smaller substrates and a higher amount of aquatic vegetation than generally available in the stream habitat. A tendency of larger trout (>10 cm) to occupy stream areas with cobble and boulder substrates and low amounts of instream vegetation distinguished their habitat use from the available habitat and from the habitat use by the smallest trout (I).

3.2. Trout densities in relation to habitat preference indices (II)

Different habitat preference curves were effective in predicting late summer densities of age-0 brown trout in among-site vs among-year comparisons. Substrate suitability index based on the summer preference curve correlated with the among-site variation of trout densities most significantly, explaining 21-74 % of this variation within the study years. The trout density among years (pooled over sampling sites) correlated, however, positively and most significantly with depth-related preference indices; preference for depth (P[d];) and the composite indices P[ds]; and P[dsv]; were best able to predict the negative effect of high discharge on trout abundance. Similarly, Nehring and Anderson (1993) found significant negative correlations between discharge and densities of brown and rainbow trout (*Oncorhynchus mykiss*) in their 13-year study in 11 Colorado streams. These results suggest that extreme flow events during the early life stages may have far-reaching effects on trout population dynamics (see also Shuler & Nehring 1994).

Habitat models can be expected to correlate closely with fish abundance only if a species' tolerable range for a habitat variable is exceeded (Shirvell 1989). This may explain why trout densities in this study were not related to the availability of suitable depths at the study sites: at most sites (ca. 80 %) P[d]; obtained values near the optimum range (0.8 - 1.0) for trout fry, which probably prevented the detection of any real associations between P[d]; and trout density. However, the negligible relationship between P[v]; and trout density does not indicate that water velocity is not an important determinant of trout distribution at a smaller scale than used in this study (3 measurements / site). Stream salmonids are known to prefer low-velocity positions adjacent to swift currents to minimize their energy expenditure, yet to have access to invertebrate drift (Fausch 1984, Hughes & Dill 1990, Hill & Grossman 1993). Thus, it is obvious that the sampling resolution for water velocity used in this study was not appropriate for describing microhabitat availability for trout (see also Heggenes 1996).

The strongest dependence of stream salmonids on suitably sized substrate particles is usually found during winter (e.g. Rimmer *et al.* 1984, Näslund 1989, Heggenes *et al.* 1993, see also paper I). In study II, sites with abundant fry in late summer often rated poorly as winter substrates. In addition, sites with the highest apparent survival obtained low indices when summer substrate curves for age-0 trout were used. These results suggest that at the onset of winter, most fry abandoned their summer habitats and moved to adjacent wintering areas where coarser substrates were available. This hypothesis is supported by many authors, who have documented considerable distances (200 m or more) moved by juvenile trout in search of suitable overwintering habitats (Cunjak & Power 1986; Chisholm *et al.* 1987).

3.3. Microhabitat selection by trout and grayling in relation to streambed complexity and flow variation (III)

The observed differences in microhabitat selection by age-0 brown trout and grayling under similar experimental conditions support Greenberg *et al.*'s (1996) contention that differences between these two species in microhabitat use result from selective rather than interactive segregation. A majority of trout preferred low-velocity refuges (see also Heggenes & Traaen 1988), whereas grayling were mostly found in swifter currents. Juvenile brown trout mostly occupied positions with a snout water velocity less than 20 cm/s (as reviewed by Heggenes 1989). Such low velocities are also used by the smallest fry of grayling, but less frequently when their length exceeds 6 cm (Sempeski & Gaudin 1995). However, since both trout and grayling selected lower velocities in winter than in summer, the potential for interspecific competition may increase with decreasing water temperature.

A homogenous substrate and a high streamflow increased the downstream movement of trout in both summer and winter. Grayling, however, were similarly affected by these factors only in winter. Larger grayling (>6 cm) are able to use much higher water velocities than their smaller conspecifics (Sempeski & Gaudin 1995, see also Valentin *et al.* 1994); i.e. neither substrate complexity nor flow level induced downstream movement of grayling (mean length 7.6 cm) in our summer experiments. While the increased downstream movement of salmonid fry under high streamflows has been repeatedly documented (e.g. Ottaway & Forrest 1983, Irvine 1986, Crisp & Hurley 1991a, b), the role of substrate complexity has been rarely addressed (but see Meyer & Griffith 1997). However, irrespective of substrate complexity, the tendency for downstream movement was higher in winter for both species, which is well in accordance with Heggenes and Traaen's (1988) finding that salmonid fry are able to withstand higher velocities at higher water temperatures.

The availability of flow refuges was much more restricted in channelized than restored flumes, and this translated to narrower preference curves with lower optimum values in channelized flumes especially for trout. Restricted availability also produced a highly clumped distribution pattern for trout in channelized flumes, especially in winter, also resulting in increased downstream movement of trout fry in winter. Overall, the observed linkage between velocity preferences and downstream movement of salmonid fry is an intraspecific analogue to the general ecological phenomenon that narrow-niched species may be more controlled by the abiotic environment than broad-niched species.

4. Intercohort competition of juvenile brown trout (IV)

Behavioural shifts in both age-0 and age-1 brown trout in the presence of the other size-class were observed in experimental flumes. Larger trout increased their activity and behaved more aggressively when together with smaller trout, whereas small trout moved less and were rarely aggressive in the presence of larger trout. These results are consistent with Bohlin's (1977) field observations that age-1 trout dominate over age-0 trout when competing for territories they both find suitable. In Greenberg *et al.*'s (1996) experiments, the habitat selection by small (ca. 10 cm) trout was not affected by the presence of larger trout (ca. 12 cm).

Size-dependent habitat use is probably a trade off between predation risk, feeding opportunities and social interactions (Hughes 1998). Daytime foraging by trout in summer would involve an increased risk of predation from visually hunting predators, especially for larger trout, possibly producing the “bigger fish-deeper habitat relationship” (see e.g. Schlosser 1987, Godin 1997, papers I and V). In winter, trout are primarily nocturnal, and selecting deep water would probably incur little, if any, benefits as regards to predator avoidance then. Nevertheless, by providing shelter from stream current, depth may still contribute importantly to habitat selection by trout even in winter (Heggenes *et al* 1993).

Fish behavior changed also seasonally, trout of both size classes being less mobile and preferring lower water velocities in winter than in summer. Both size classes increased their use of instream cover in winter. Such wintertime aggregation of trout to low-velocity habitats is well documented in previous studies (e.g. Cunjak & Power 1986, Heggenes & Saltveit 1990, see also papers I and III). When both size classes were present, only small trout changed their use of water velocities and cover, whereas large trout did not. The results indicate that intercohort competition may indeed cause spatial segregation among size groups of brown trout.

Contrary to some recent findings reporting diel variation in the behavior of salmonid fish (e.g. Valdimarsson *et al.* 1997, Metcalfe *et al.* 1998), no diel periodicity was observed in the behavior or microhabitat use of brown trout regardless of season. Heggenes *et al.* (1993) found brown trout to be nocturnally active at low water temperatures during winter, whereas in daytime trout were concealed within the interstitial spaces of the stream substrates. The lack of diel periodicity may be due to the use of hatchery fish in this experiment; hatchery fish are reported to be day-active irrespective of water temperature (Pirhonen 1998). Furthermore, nocturnal behavior of salmonids is at least partly a means of avoiding diurnal predators (Valdimarsson & Metcalfe 1998), which hatchery fish are unfamiliar with.

3.5. Spatial relations between lotic fish, benthic prey and the stream habitat (V)

Overlay maps obtained from field mapping surveys show the tendency of large trout (10-15 cm) to occupy deeper stream areas with coarser substrates than the ones typically used by smaller trout (5-10 cm). Spatial relations between fish and their prey was detected for only Arctic bullhead. Large bullhead (> 4 cm) aggregated in areas providing the highest abundance of benthos, especially semisessile invertebrates (larval stages of black flies and filter-feeding caddisflies). For predominantly drift-feeding trout (Allan 1981, Grant & Noakes 1986, McIntosh & Townsend 1995), benthic samples may give a somewhat biased view of prey availability. However, if spatial associations exist, they should be found between fish and semisessile prey (see Sih 1984), because mobile prey types may continuously shift their distribution in relation to local predation pressure (Tikkanen *et al.* 1994, Forrester 1995).

High-density patches of trout and bullhead were in separate sections of the sample reach. The spatial interactions between these two species appear to be highly scale-dependent: inhibition at small scales (intraspecific aggregations) vs coherence at larger scales (whole stream sections; A. Mäki-Petäys, unpubl.) Thus, the near-absence of trout from high density prey patches may be explained by interference between trout and bullhead. In many occasions bullhead are suggested to compete for food with juvenile salmonids (e.g. Andreasson 1971, Mason & Machidori 1976,

Andersson *et al.* 1986). In addition, Gaudin and Caillere (1990) suggested that juvenile brown trout avoid areas with high density of bullhead (*Cottus gobio* L.).

Semivariograms showed spatial dependence in the data for water depth and density of semisessile invertebrates, especially at lags shorter than 2 meters. There was probably, however, also some dependence in invertebrate densities at scales below the smallest sampling distance (1.0 m). It is impossible to sample fish and benthic macro invertebrates with the same accuracy using the same sampling grid. Thus, while the measurements of this study were accurate for fish distributions, a denser grid should have been used for benthic invertebrates. Overall, the GIS-assisted (Geographical Information System) approach described in this study combined with standard geostatistical tools and statistical modelling may prove valuable for determining appropriate sampling scales for future studies of fish habitat in relation to the distribution of benthic and drifting prey (see also Muotka & Penttinen 1994).

Chapter 4. Implications for ecologically-based fisheries management in boreal streams

It is not surprising that habitat-hydraulic modeling has won increasing popularity among fisheries biologists when selecting management strategies for stream salmonids. By estimating the amount of habitat suitable for fish in different phases of their life span, such modeling facilitates evaluation of the potential of a stream reach for fish production. Recently, however, many authors have underlined the flexibility and dynamic nature of fish habitat use, and habitat suitability criteria and modeling based on single snapshot studies have met with limited success (e.g. Orth 1987, Heggenes 1996). Since seasonal and ontogenetic changes in habitat use and preference by juvenile salmonids were evident in all studies included in this thesis, I suggest that the minimum requirement for increasing the biological realism of instream models is that the size structure of the fish population under study and the seasonality of resource availability and use be incorporated into the models.

Stream habitat and fish abundance will be tightly linked only if habitat availability is the prime determinant of fish density (e.g. Milner *et al.* 1985). Many other factors (see paper I) can contribute in determining fish abundance, potentially masking any habitat-fish density relationships (Milner *et al.* 1985, Orth 1987, Heggenes 1996). As with any model, the predictive accuracy and realism of the physical habitat model output is limited by the user's understanding of the underlying assumptions and limitations inherent in the model, in addition to the biological knowledge and expertise incorporated into the modeling and decision-making process (Huusko & Yrjänä 1997; see also Gan & McMahon 1990, Heggenes 1996). Unfortunately, the construction of independent habitat preference criteria for all of the various aspects of fish habitat use will not be logistically feasible. However, when interpreting the output of habitat-hydraulic models in relation to fish abundance, we should at the very least be able to estimate the extent to which biotic interactions and other factors, not included in the model, may modify fish habitat selection. The correlative nature of most field studies may, however, prevent insight into the actual mechanisms underlying the responses of fish populations to variations in stream habitat conditions. Consequently, it is advisable to test the detected patterns by manipulative experiments under controlled conditions (III, IV; see also Fausch 1992). Furthermore, a higher sampling resolution and a multi-scale sampling design may provide us with a better

understanding of the mechanistic links between fish distribution and the spatial heterogeneity of the stream habitat. Finally, because experimental studies in indoor flumes mainly focus on small scale behavioral responses of young fish, inevitably under artificial conditions, a larger scale experimental system consisting of replicable seminatural stream units is urgently needed to facilitate habitat studies at scales relevant to all size groups of fish.

Assessing the productive potential of a stream in relation to species-specific habitat requirements and identifying key factors underlying 'bottleneck' periods for fish populations are the cornerstones for effective management of threatened salmonid populations (see Fausch *et al.* 1988, Thorfve & Carlstein 1998). In boreal areas, harsh winter conditions are critical for salmonid production (II and III), and as pointed out by Cunjak (1996), it is "...not until we consider winter as part of the field season rather than a time to work up summer data, will we be able to effectively conserve and manage fish habitat." Wintering trout often shelter among the interstitial spaces of coarse substrates (e.g. Heggenes *et al.* 1993), and to facilitate the survival of juvenile trout through winter, stream management programs need to ensure that such spaces are abundantly available in trout wintering areas, and that seasonal habitat shifts between summer nursery areas and overwintering habitats are unrestricted (see Cunjak 1996). Therefore, winter preference curves should be used when habitat-hydraulic modeling is applied in areas where winter conditions may limit habitat availability for juvenile trout.

Habitat preference curves developed for brown trout in this thesis, combined with habitat-hydraulic modelling, should facilitate the estimation of habitat suitability for juvenile trout during different phases of their life in boreal streams. This will help us elaborate stream restoration schemes that take the habitat requirements of juvenile trout into serious consideration. Then, management efforts could be focused on preventing stream habitat deterioration, such as loss of trout nursery areas, instead of merely compensating losses in trout production by stocking. This approach, i.e. assessment of the amount of habitat suitable for trout in a stream, should also be helpful when determining appropriate stocking densities for different streams. From the management point of view, it is important to notice that microhabitats selected by trout fry differed from those selected by grayling. Both these species are important game fish and often live sympatrically in many northern streams. If the goal of a rehabilitation program is to provide suitable living conditions for both trout and grayling, species-specific preference curves should be used in habitat-hydraulic modeling. Since such detailed knowledge is often lacking, however, the most prudent option may often be simply to provide a broad diversity of microhabitats to ensure suitable living conditions for various life-stages of most stream salmonids (see also Greenberg *et al.* 1996, Huusko & Yrjänä 1997). A heterogeneous stream habitat is a worthy goal also because it provides scope for seasonal and ontogenetic variation in microhabitat selection by both trout and grayling.

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Appendix H

AQUATIC WINTER HABITAT MONITORING STUDY

and

ASSESSMENT OF THE INTERACTIONS BETWEEN STREAM DISCHARGE AQUATIC HABITAT. AND THE TROUT POPULATION OF THE DOLORES RIVER BELOW MCPHEE DAM

By

R. Barry Nehring
Wildlife Researcher
Colorado Division of Wildlife
Montrose, Colorado 81401

For

U.S. Department of the Interior
Bureau of Reclamation
Durango Projects Office
Durango, Colorado 81302

Contract Number

1 —FC—40— 10460

May 24, 1991

INTRODUCTION

This study was entered into by the Colorado Division of Wildlife (CDOW) upon request by the United States Bureau of Reclamation-Durango Projects Office (USBR-DPO). Due to the severe drought conditions affecting southwestern Colorado since 1988, the flows in the Dolores River below McPhee Dam were reduced to 20 cubic feet per second (cfs) in early-March 1990. Unusually warm weather in May and June 1990, together with extremely low discharge, resulted in rapidly rising water temperatures between the dam and Bradfield Bridge.

Concern was expressed (by both public resource agencies and private citizen groups) that the entire trout population in the Dolores River tailrace fishery might be lost to disease, stress, and thermal shock. In an attempt to avoid this potential calamity, the USBR-DPO increased the discharge to 50 cfs by mid-June 1990, and maintained that flow through the first week in September, when flows were again reduced to 31 cfs. The purpose of the winter monitoring study (under Cooperative Agreement No. 1 FC-40-10460) was to monitor the extent of the winter ice build—up under the 31 cfs flow regime and evaluate the effects of icing on the fishery and aquatic invertebrate populations in the Dolores River between McPhee Dam and Bradfield Bridge.

METHODS

Monitoring of Ice Development

The development of icing in the Dolores River was monitored at least twice weekly from early-December 1990, through March 1, 1991. Photographic stations were set up at six different locations along the Dolores River corridor below McPhee Dam and Bradfield Bridge. The upper station was located at the McPhee Dam stilling basin. The five remaining stations were located at approximately river mile(s) 0.5, 3, 6, 9, and 12 downstream from the dam. One upstream and one downstream photograph was taken at each location on each photograph day. The photos were developed, labeled (by station, view, and date), and compiled in a photo album.

The photographs were reviewed for a visual estimate of the amount of ice cover on each day. The amount of ice cover was expressed as a percentage of ice cover, i.e., 0 = none and 100 = total ice cover. This information is summarized by station and date for the entire winter monitoring period in Table 1, except for river mile 0.5 where no ice developed. Thus it was excluded. The development, extent, and fluctuation in the percentage ice cover at each station was correlated with ambient air temperature (from the weather station at Bradfield Bridge- 12 miles downstream from McPhee Dam) and with water temperatures at McPhee Dam, river mile 6, and river mile 12. Water temperatures were monitored using RYAN TEMP MENTOR temperature recording thermographs. Air temperatures (on photograph days) at the Brad^lfield Bridge and water temperature data for December 1990 through March 5, 1991, are given in Table 2.

Aquatic Invertebrate Sampling Regime

Aquatic invertebrate samples were collected once a month, from December 1990, through March 1991, at three sites on the Dolores River within the winter study area. These sites were located at river mile(s) 0.5, 6, and 12, corresponding closely to the three water temperature monitoring stations. Three Surber (1 ft²) samples were collected at each of the three bites on each of four sampling dates. A list of the aquatic invertebrate taxa sampled with mean numbers/ft² for each monthly sample are shown in Tables 3 through 5.

Fish Sampling and Observations

Trout population monitoring and evaluation was completed by comparing the trout population density and biomass estimates from October 1990, with those from April 1991, as well as previous years. Population estimates have been conducted each year since 1987 using the Seber-LeCren two pass removal method (Seber and Le Cren 1967). Biomass estimates were computed using a length-weight regression (LWR) analysis derived from fish length (cm) and weight (g) data pairs taken during electrofishing surveys.

Age and growth analyses of the populations have also been completed using scale aging and back-calculated length analysis. This information is contained in Table I-i. in the Appendix of the report.

Approximately 60 trout (? 20 cm) were tagged at three study sites in April 1991. Visual implant (VI) tags were inserted under the skin behind the right eye with a tag insertion needle. This equipment is manufactured and marketed by Northwest Marine Technology, Olympia, WA. These fish were tagged and released to aid in future age, growth, and movement studies. A scale sample was collected from each tagged fish so that both the length and the age of the fish is known (for future reference). Data on the trout implanted with VI tags is given in Table 1-2 in the Appendix of the Report.

Underwater fish observations were conducted in January and February 1991 , using dry suits, snorkels, masks, and winter diving gloves. Both day and night snorkeling observations were carried out to assess differences in behavior between daylight and darkness'. Observations and measurements on individual fish and fish habitat characteristics included the following measurements:

- 1) Fish length (by size category)
- 2) Species
- 3) Fish activity
 - a. random swimming
 - b. stationary swimming
 - c. feeding
 - d. resting
- 4) Water column depth
- 5) Mean water column velocity
- 6) Nose velocity
- 7) Depth strata of the fish (upper, middle, lower third of water column)

- 8) Cover Type
 - a. no cover
 - b. Visual isolation
 - c. Velocity shelter
 - d. combination (object cover/visual isolation)
- 9) Size of Object Cover and Distance to Object Cover
 - a. less than 1 foot (diameter & distance,,
 - b. 1 to 2 feet (diameter and distance)
 - c. 2 - 4 feet (diameter and distance)
 - d. greater than 4 feet (diameter and distance)

These observations were made to assess potential changes and differences of habitat use as well as overall body condition of the trout observed during the winter months.

RESULTS

The information in Table 1 is an approximation of the percent ice cover upstream and downstream of each photo station on each observation day. The data indicates that dramatic increases in both the percentage of ice cover and river mileage affected began around December 21, 1990. This corresponds to the beginning of a two week period when overnight air temperatures (at the Bradfield Bridge) ranged from -20 to - 27 F for 11 of 14 days. Daytime highs rarely reached 32 F during this period. Information on air and water temperatures at the various monitoring stations is presented in Table 2.

The severe cold snap resulted in a 100 percent ice cover over all pool habitats and near total ice cover even in the riffle areas at all points more than 6 miles below McPhee Dam. These areas remained essentially ice—covered through mid—February 1991. Overnight minimum air temperatures during January 1991 were below 0 F for 17 of 31 nights. It was not until February 7, 1991, that a. overnight low air temperatures stayed consistently above 0 F and daytime high air temperatures were consistently above the freezing (32 F) mark. On many days after February 1, 1991, the maximum daily air temperatures reached into the high 40s.

The ice cover on the Dolores River (below the dam) began receding rapidly at river miles 6 and 9 between February 15 and 22, 1991. This was clearly the result of warming of the water during the daylight hours by solar insulation rather than an increase in temperature of water released from McPhee Dam. Water temperatures in the stilling basin at McPhee Reservoir remained consistently between 37.5 and 38.8 F throughout January and February 1991. However, water temperatures at river mile 6.6 (Temp Mentor site) began increasing above 32 F on February 6, 1991, and reached daytime maximums of 40 - 44 F every day except two after February 14th. This period corresponds well with the time when overnight low air temperatures at Bradfield Bridge consistently remained above +10 F.

Table 1 Approximate amount of ice cover (percent of channel frozen) at various sites on the Dolores River, in river miles below McPhee Dam, December 1990 — March 1991.

Date Mo/d	Station Distances in River Miles below McPhee Dam							
	12.2 miles		9 miles		6 miles		3 miles	
	Down	Up	Down	Up	Down	Up	Down	Up
12/3	75	50	---	---	---	---	---	---
12/6	75	50	25	50	90	75	0	0
12/11	75	50	25	50	90	75	0	0
12/14	75	50	25	25	0	10	0	0
12/18	90	50	25	25	0	50	0	0
12/21	100	50	75	75	100	95	20	20
12/24	100	75	75	90	100	100	90	90
12/27	100	90	90	95	100	100	100	95
1/1	100	95	95	95	100	100	50	60
1/4	100	90	95	95	100	100	20	20
1/8	100	90	95	95	100	100	10	10
1/11	100	90	95	95	100	100	10	10
1/18	100	95	95	95	100	100	10	10
1/22	100	95	90	95	100	100	10	20
1/25	100	95	90	90	100	80	10	10
1/29	100	95	80	90	80	100	10	0
2/8	100	95	80	90	100	75	5	0
2/15	100	95	80	90	10	20	5	0
2/22	100	75	20	20	0	0	0	0
2/26	75	25	10	10	0	0	0	0
3/1	20	0	0	0	0	0	0	0
3/5	10	0	0	0	0	0	0	0

Ice out at the Bradfield Bridge was not complete until March 1, 1991. This corresponds to the Temp Mentor record at the bridge which indicates daytime high water temperatures did not reach 33 F until February 26, 1991, and 40 F on March 1, 1991.

Aquatic Invertebrate Samples

The aquatic invertebrate samples for the Dolores River below McPhee Reservoir (Tables 3 through 5) indicate the general trends expected for tailrace lotic (stream) environments. The species diversity was poorest at the station nearest the dam with only 16 taxa (macro-invertebrate groups and/or species) being found during the winter sampling period. At river mile 6 a total of 26 taxa were found. The greatest diversity occurred at river mile 12 (near Bradfield Bridge) with 30 taxa being collected and identified. In contrast, 27 taxa were collected at one site on the Gunnison River (below Crystal Dam) in April 1981. At a second site 24 taxa were collected, and 15 taxa at a third site during the same sampling period (Bio/West 1981). More than 50% of the taxa were common to both the Dolores and Gunnison rivers.

Table 2. Periodic air and water temperature (Fahrenheit) information for the Dolores River Corridor below McPhee Dam to Bradfield Bridge, December 1990 - March 1991.

Date Mo/d	Bradfield Bridge Air Temperature		Daily Water Temperatures (F)		
	High	Low	McPhee	Mile 6.6	Mile 12
12/3	37	-5	42.5	33.2	32
12/6	36	+5	41.5	32.0	---
12/11	50	+7.5	41.0	34.0	---
12/14	38	34	40.8	34.0	---
12/18	31	-3	40.0	32.0	31.8
12/21	16	-26	37.5	32	31.8
12/24	13	-26	38.7	32	31.8
12/27	31	-25	39	32	31.8
1/1	34	-23	38.5	32	31.8
1/4	35	+27	38.5	32	31.8
1/8	41	+29	38.5	32	31.8
1/11	38	-5	38.0	32	31.8
1/18	43	0	38.3	32	31.8
1/22	32	-21	38.3	32	31.8
1/25	35	-10	38.3	32	31.8
1/29	28	+15	38.3	32	31.8
2/8	43	+16	38.3	35.5	31.9
2/15	55	+13	38.3	37.6	31.9
2/22	51	+17	38.3	37.5	31.9
2/25	46	+10	38.3	38.5	32.7
3/1	44	+22	38.3	39.0	40.0
3/5	46	+27	38.5	40.5	41.0

Silt tolerant species (especially Trichoptera) like net— spinning and filter-feeding *Hydropsyche* sp., which feed on detrital material, were found in larger numbers at the most downstream station, at intermediate densities at station 6, and in the lowest densities at station 0.5 (Tables 3 — 5). This is a common phenomenon in stream ecosystems (Hynes 1970).

Table 3. Dolores River winter 1990/1991 macroinvertebrate sampling, approximately 0.5 miles below McPhee Dam.

Taxonomic Description	Date	Numbers/ft.			
		12/4/90	1/3/91	2/12/91	3/19/91
Class/Order/Family/Genus/Sp.					
Oligochaeta		6.6	19	14	34.3
Hydracarina//Sperchon		1.6	7	1.6	0.33
Coleoptera/Elmidae/Zaitzevia		----	----	0.33	----
Ephemeroptera//Baetis		105	230	273	313
Ameletus		0.66	----	----	----
Paraleptophlebia		27.5	26.6	11.6	2.6
Serratella		2.3	4	2	4.6
Plecoptera/Nomouridae/Capnia		0.33	----	----	----
Allocaenia		----	0.33	----	----
Trichoptera//Hydropsyche		1	2.6	1	1.3
Ochrotrichia		4.3*	41	11	5.6
Rhyacophila		----	----	0.33	----
Diptera/Simuliidae		1.3	15.3	52.6	81.6
Chironomidae		201	223	266	505
Tipulidae/Dicranota		----	1	----	2
Anthomyiidae/Limnophora		9	8.3	3	7

* Species present in December 1990 samples, but went unrecognized at that time.

Table 4. Dolores River winter 1990/1991 macroinvertebrate sampling, approximately 6 miles below McPhee Dam.

Taxonomic Description	Date	Numbers/ft.			
		12/4/90	1/3/91	2/12/91	3/19/91
Class/Order/Family/Genus/Sp.					
Oligochaeta		4.3	0.6	12.3	3.6
Gastropoda//Ferrissia		2.6	2	2.6	3.6
Turbellaria//Tricladida		*	15.6	27	23
Anchipoda//Hyalis azteca		0.33	0.33	----	----
Coleoptera/Elmidae/Zaitzevia		12	18	21.3	12.6
Heterolimnias		0.33	2	----	1
Narpus		----	----	0.66	----
Ephemeroptera//Baetis		7.3	76	107	136
Epeorus (Iron)		0.33	3	2	1.6
Heptagenia		----	0.33	----	----
Ameletus		0.33	----	----	----
Paraleptophlebia		1	10	11.6	4
Serratella		----	----	0.33	----
Tricorythodes		1	4.6	1	2
Plecoptera//Claassenia sabulosa		1	0.66	0.33	0.33
Isoperla		2.3	4.6	3	8.6
Isogenus		----	----	----	0.33
Trichoptera//Hydropsyche		71	112	97	216
Polycentropus		8	0.33	0.66	1
Perlepis		----	1.3	0.66	2.6
Psychorenia (spelling?)		----	0.33	----	----
Diptera/Simuliidae		34.6	19	472	60.3
Chironomidae		4.6	14.6	12	23
Tipulidae/Dicranota		0.33	0.33	0.33	1
Hexatoma		1	1	2.3	1.6
Anthomyiidae/Limnophora		----	----	0.33	----

* Species present in December 1990 samples, but went unrecognized at that time.

Table 5. Dolores River winter 1990/1991 macroinvertebrate sampling, near Bradfield Bridge, 12 miles below McPhee Dam.

Taxonomic Description	Date	Number of Individuals			
		12/4/90	1/3/91	2/12/91	3/19/91
Class/Order/Family/Genus/Sp.					
Oligochaeta		4	1.6	2	1.6
Decapoda		0.33	0.33	----	----
Hydracarina///Sperchon		----	----	0.66	0.33
Gastropoda///Ferrissia		1.6	6.3	1.6	0.66
Anhipoda///Hyaloleia azteca		----	0.66	0.66	----
Coleoptera///Zaitzevia		30	29	19	31
Heterolimnius		2	1.3	2.3	1.3
Narpus		0.33	----	----	----
Lepidoptera///Parargyractis		*	----	----	----
Odonata///Hyponeura lugens		----	0.33	----	0.33
Ephemeroptera//Baetis		15.3	75	194	232
Epeorus (Iron)		----	1.5	2	----
Paraleptophlebia		16.3	17.6	19.6	5
Heptagenia		0.33	----	----	----
Tricorhythodes		1.6	12.6	9.6	7.6
Plecoptera//Claassenia sabulosa		3	2	3	2.3
Isoperla		5.6	4.3	5.3	2
Isogenus		----	----	----	0.33
Trichoptera//Hydropsyche		248	158	107	70
Ochrotrichia		*	21	18	8
Polycentropus		----	1.3	1.3	0.33
Glossosoma		----	----	----	4.6
Psychomyia		----	----	----	0.33
Oecetis		7	15	8	1.3
Diptera/Simuliidae		45	6	45	937
Chironomidae		55	62	130	231
Tipulidae/Pedecia		----	----	----	0.33
Tipulidae/Dicranota		----	----	0.33	----
Anthomyiidae/Limnophora		----	1.6	5	0.33
Stratiomyidae		----	0.33	----	----

* Species present in December 1990 samples, but went unrecognized at that time.

Table 6. Dolores trout population River (below McPhee Dam) Seber-LeCren two pass removal and biomass estimates for all trout ~ 15 cm, 1987-1991.

S	October 1987		October 1988		Septemb 1989		October 1990		April 1991	
	N/Ha	Kg/Ha	N/Ha	Kg/Ha	N/Ha	Kg/Ha	N/Ha	Kg/Ha	N/Ha	Kg/Ha
IFIM Site (3B-20 - River Mile 0.5)										
Bn	104	18.4	----	-----	64	10.3	67	14.6	63	8.7
Rb	---	-----	---	-----	89	33.0	73	32.8	50	19.7
Ct	23	10.2	----	-----	30	11.7	16	17.2	5	2.1
Tl	145	35.0	----	-----	183	55.0	142	64.6	90	30.5
Ferris Canyon Site (3B-15 - River Mile 6)										
Bn	43	7.5	101	34.5	116	34.2	81	25.7	44	12.8
Rb	----	-----	52	5.8	82	31.2	55	23.3	15	6.3
Ct	----	-----	20	9.1	10	7.1	2	2.3	89	21.6
Tl	92	24.5	183	51.4	208	72.5	133	51.3	93	46.9
Rock Stockpile Area (3B-10 - River Mile 10)										
Bn	91	13.7	167	42.8	99	16.1	79	10.9	55	6.3
Rb	14	5.3	52	9.0	82	13.0	55	5.5	34	10.5
Ct	----	-----	99	15.9	20	5.5	0	0	57	2.8
Tl	68	21.7	318	66.7	201	34.6	131	17.4	130	19.6

† S=species; Bn=brown; Rb=rainbows; Ct=cutthroat; Tl=total trout

Table 7. Dolores River (below McPhee Dam) Seber-LeCren two pass removal trout population and biomass estimates for all trout 2 15 cm, 1987—1991. Species (size cm) October 1988 September 1989 October 1990 April 1991

Species (size cm)	October 1988		Septemb 1989		October 1990		April 1991	
	N/Ha	Kg/Ha	N/Ha	Kg/Ha	N/Ha	Kg/Ha	N/Ha	Kg/Ha
IFIM Site (3B-20 - River Mile 0.5)								
Brown ≥40	0	0	2	0.9	9	5.8	7	4.6
Brown ≥20 & ≤40	52	12.1	26	7.1	47	11.8	14	1.9
Rainbow ≥40	5	5.5	22	20.2	34	32.8	22	17.2
Rbw ≥20 & ≤40	21	7.3	33	14.3	17	3.9	16	3.4
Cutthroat ≥40	7	8.6	8	7.5	18	20.0	3	2.0
Cutts ≥20 & ≤40	5	1.0	9	4.0	0	0	0	0
Total Trout ≥40	12	14.1	31	28.6	56	58.6	30	23.8
Ttl Trt ≥20 & ≤40	78	20.4	70	25.4	54	15.7	23	5.4
Ferris Canyon Site (3B-15 - River Mile 6)								
Brown ≥40	11	8.1	29	20.6	29	20.2	10	14.7
Brown ≥20 & ≤40	90	25.2	96	24.9	58	12.8	23	4.0
Rainbow ≥40	13	10.8	21	14.7	20	16.1	5	4.2
Rbw ≥20 & ≤40	44	15.9	53	13.9	30	6.8	5	1.7
Cutthroat ≥40	7	6.0	5	4.7	2	2.2	24	18.9
Cutts ≥20 & ≤40	0	0	3	1.4	0	0	0	0
Total Trout ≥40	25	24.8	83	40.0	50	39.2	33	37.9
Ttl Trt ≥20 & ≤40	112	41.1	149	40.1	86	19.6	30	5.7
Rock Stockpile Area (3B-10 - River Mile 10)								
Brown ≥40	7	4.3	4	1.9	0	0	2	1.6
Brown ≥20 & ≤40	144	31.3	88	14.4	78	12.7	25	3.5
Rainbow ≥40	6	5.1	4	3.4	2	2.0	10	8.4
Rbw ≥20 & ≤40	12	4.6	58	10.0	43	5.9	15	1.7
Cutthroat ≥40	6	6.9	2	2.4	0	0	0	0
Cutts ≥20 & ≤40	35	5.8	10	2.8	0	0	0	0
Total Trout ≥40	18	16.3	10	3.4	2	2.0	7	10.0
Ttl Trt ≥20 & ≤40	182	42.7	154	27.1	127	18.6	32	5.1

Table 8. Stream flow (cfs) summary for the Dolores River below McPhee Dam from April 1984 through April 1991.

Year	Spill Period	Spill Range	Minimum Flow (MF) Ranges	MF Time Period
1984	4/10-7/4/84	500-3900	64-700	7/8/84-3/13/85
1985	3/14-6/27/85	500-3700	50-100	7/11/85-3/28/86
1986	4/1-6/27/86	500-4400	78-600	7/1/86-3/17/87
1987	3/10-6/30/87	500-3300	40-80	9/15/87-4/27/88
1988	4/28-6/8/88	500-1200	78-80	6/17/88-3/30/89
1989	4/17-5/14/89	500-1000	78-200	5/23/89-3/1/90
1990	No Spill	-----	19-21	3/6/90-6/14/90
1990	No Spill	-	29-63	6/15/90-9/3/90
1990	No Spill	-----	29-39	9/5/90-4/30/91

Winter flows (1986/87) varied from 125 to 175 cfs. Flows varied from 120 - 270 cfs from 7/1/87—9/13/87

The trout population density and biomass data in Tables 6 and 7, taken together with the flow data summarized in Table 8, indicate a strong cause and effect relationship between decreasing stream flows and decreasing trout density and biomass since October 1988. It stands to reason that the effects of reduced flows and elevated summer water temperatures would be felt at an earlier time and with more pronounced effect as distance (in river miles) below McPhee Dam increased. That is what has happened.

Numerical density and biomass peaked in the fall 1988 at the Rock Stockpile Area (at approximately river mile 10). Total trout density estimates (N/ha) for 1988, 1989, 1990, and April 1991 were 318, 201, 131, and 130, respectively. Total trout density (kg/ha) estimates for the same period(s) were 66.7, 34.6, 17.4, and 19.6; again a rather sustained and precipitous decline after 1988. Thus, total trout density and biomass at river mile 10 have declined 59% and 71%, respectively, between October 1988 and April 1991 at the most downstream station.

Farther upstream at river mile 6 (Ferris Canyon site) trout population density and biomass peaked in 1989. Most of the positive change (between 1988 and 1989) occurred in the rainbow component of the trout population. Brown trout density and biomass remained relatively stable during this period, while Snake River cutthroat (SRC) trout density and biomass declined. Total trout density (N/ha) at this site for the fall of 1988, 1989, 1990, and April 1991 was 183, 208, 133, and 93, respectively. Similarly, total trout biomass (kg/ha) for the same period was 51.4, 72.5, 51.3, and 46.9. Total trout density and biomass decreased at the Ferris Canyon site by 55% and 35%, respectively, between October 1989 and April 1991.

Finally, at the upstream station (IFIM study site at river mile 0.5) total trout density (n/ha) peaked in 1989 and biomass (kg/ha) peaked in October 1990. No population estimate was completed in October 1988 due to a severe drop in stream discharge during electroshocking

operations, thereby biasing any results. However, data is available for October 1987, as well as the fall of 1989, 1990, and April 1991. Total trout density (n/ha) for these time periods was 145, 183, 142, and 90. Similarly, total trout biomass (kg/ha) for the same period was 35.0, 55.0, **64.6**, and 30.5, respectively. Thus, total trout density declined by 51% between October 1989 and April 1991. Similarly, total trout biomass declined 53% between October 1990 and April 1991.

It is quite likely that both density and biomass in October 1990 (at the upstream study site) were artificially high due to upstream migration of trout during the summer of 1990. This was probably stimulated by a combination of warm water temperatures, low flows, and habitat degradation due to excessive silt accumulation in downstream areas. Recent studies with radio tagged trout in Michigan and Wisconsin indicate trout will migrate long distances to avoid thermal stress (Clapp et al 1990; Wisconsin DNR, personal communication),

Trout age and Growth

The age, growth, and back—calculated length analyses for brown, rainbow, and SRC trout are contained in Table I-1 in the Appendix. This information indicates that all three species of trout grow quite rapidly in the Dolores River below McPhee Dam. Most trout range in size from 30 to 40 cm (12'—16") at age 3, with the largest trout reaching 43-44 cm (17 inches) at that age. The average length of most cohorts (year classes) exceed 40.7 cm (16 inches) by age 4. This compares favorably with rainbow and brown trout growth rates in the Gunnison River below the Curecanti USBR Project. (Nehring 1988).

Trout Tagging

Most of the trout 20 cm in size captured during electrofishing operations during April 1991 were marked with visual implant (VI) tags. Twenty-one trout (five rainbow, one cutthroat, and 16 browns) were tagged at the downstream station (river mile 10). Twenty-two trout (four rainbow, three cutthroat, and 15 browns) were VI tagged at the middle station (river mile 6). Fourteen trout (10 rainbow, one cutthroat, and three browns) were VI tagged at the upper station (river mile 0.5). Scale samples were taken from each of these fish to determine their age. The tagging information (tag number, species, and total length — cm) by station is presented in Appendix Table 1—2.

Winger Snorkeling Observations

Ninety-seven individual underwater observations on rainbow, brown, and cutthroat trout were made during January and February 1991. Snorkeling observations were conducted on January 31, February 1, 26, 27, and 28. Fourteen of the 97 observations were obtained during daylight hours, while 83 were collected at night (7-11 PM). The only trout observed during daylight hours were very large individuals (240 cm). In contrast, the vast majority of trout observed during the night time observation period(s) were smaller trout, i.e., 6 — 30 cm in length. The smaller trout, apparently nocturnal during the winter months, remained hidden in cryptic areas of the stream during daylight hours.

Other investigators have documented this change in behavior among salmonids during the fall/winter period (Campbell and Neuner 1985). This shift to a behavior pattern characterized by cryptic inactivity during daylight hours and nocturnal activity is a gradual process that begins to occur when fall water temperatures begin dropping below 8 C. Once water temperatures drop below 3 C, trout 150 mm in length and smaller were rarely encountered during snorkeling observation forays during daylight hours (Campbell and Neuner 1985).

The measurements made on winter habitat use by the trout in the Dolores River for water depth, mean column velocity, kind nose velocity are summarized in Table 1—3 in the Appendix. Perhaps the only conclusive information contained in the data is that size category 2 (8-12 inches) and 3 (2 12 inches) trout (regardless of species) use water at least 2 feet in depth or greater. Size category zero and one (4 inches and 4-8 inches) trout were found in water less than 2 feet deep. Whether this was by choice or due to intraspecific and/or interspecific competition could not be determined. The study by Camobell and Neuner (1985) leads me to conclude that it is most probably by choice and an innate behavioral response by smaller size trout that minimizes competition.

All of the trout observed during the winter period appeared to be in good condition, with no outward signs of stress or disease from low water conditions, cold water temperatures, and/or icing. Length/weight regression (LWR) coefficients for trout collected during the electrofishing surveys in April 1991 were definitely in the normal range. The beta coefficients for the LWR for brown, rainbow, and cutthroat trout were 2.92, 2.84, and 3.11, respectively. A beta coefficient of 3.0 is about the mid—point of the normal range.

That does not mean there has been no stress on the trout population over the past two years. On the contrary, beta coefficients calculated for trout length/weight regressions for the Dolores River trout in the fall of 1989 were 2.73, 3.22, and 3.23 for brown, rainbow, and cutthroat trout, respectively. Thus, the body condition of the rainbow and cutthroat trout (at that time) was much better than in the spring of 1991. Therefore, it is clear that the decrease in weight (g) per unit length (cm) between the fall of 1989 and April 1991 is real. Other studies have documented weight loss and decrease in body condition among salmonids during the winter months (Cunjak 1988).

This decrease in body condition, however, is probably due to the cumulative impacts of poorer habitat conditions brought on by the decreased flows in the river between March and November 1990 and less to the 30 cfs between December 1990 and April 1991. The winter period, when water temperatures are near 0 C (32 F), is a time when metabolic activity is minimal and low water conditions are less stressful. The spring—fall period (when water temperatures are in the 50 - 68 F range) is the time of maximum metabolic activity. Low water conditions during this period are going to be far more stressful. The negative aspects of low flow during the summer months have a compounding effect. The food producing , capability of the stream is impacted, habitat availability is reduced, density of the trout is artificially increased, disease vectors (such as fungal spores) are concentrated, and elevated water temperatures (above 11—13 C or 52-55 F) result in lower metabolic efficiency for trout (Elliott 1975 a,b). Undoubtedly, some of these negative effects are synergistic rather than additive, compounding the stress factor(s) considerably.

No dead or dying trout were observed during the winter snorkeling studies. Snorkeling observations on January 31 and February 1 were confined to an area within 2 miles of McPhee Dam while the river was ice covered downstream from that point. However, snorkeling observations in late February were conducted up to 10-11 miles downstream of McPhee Dam just a few days after ice-out. Had there been a significant die-off of trout due to icing, trout carcasses would definitely have been visible asunder water visibility was greater than 15 feet for both daytime and nighttime snorkeling. Even more important, it was readily apparent from the number of young trout (6 — 20 cm) observed during the nighttime hours that the smaller size groups survived the winter very well. Rainbow and brown trout in size classes zero (~ 10 cm) and one (10 — 20 cm) comprised 51% of all the underwater snorkeling observations. These two size classes are the trout that burrow into the interstitial spaces in the substrate in riffle areas during the winter (Campbell and Neuner 1985). It is in the riffle areas that young trout would be most vulnerable to the effects of (1) anchor ice build—up and/or (2) being crushed or frozen in the substrate if sheet ice development in riffle areas had been a major problem.

Young rainbow, brown, and cutthroat trout (20 cm total length) were observed in abundance during both the winter snorkeling and the April 1991 electrofishing operation. This gives strong credence to the contention that trout mortality effects directly linked to ice development in the Dolores River channel below McPhee Dam this past winter were probably minimal.

Brown Trout spawning Success/Fry Recruitment

The brown trout population in the Dolores River between McPhee Dam and Bradfield Bridge is currently a self—sustaining population. No brown trout fingerlings have been stocked in this section of the river since July 1988. The data in Table 9 indicate that brown trout fry have been collected in ever—increasing numbers since October 1988. While total trout density and biomass has been in a steady decline since the fall of 1988 (at river mile 10) and *I.* fall 1989 (at river mile 6), brown trout reproductive success has nonetheless been on the increase. The number of YOY brown trout collected in April 1991 is 400% higher than October 1988. Thus, it is possible to conclude that spawning success has not been seriously impacted by the extremely low water conditions in this section of the river since March 1990.

Table 9. Brown trout fry. samples (5 — 15 cm), Dolores River below McPhee Dam, 1988 - 1991.

Station Description (CDOW I.D. & River Mile)	Sampling Period			
	10/1988	9/1989	10/1990	4/1991
IFIM Site (3B-20 River mile 0.5)	6	4	10	10
Ferris Canyon Site (3B-15 @ River Mile 6)	3	0	19	25
Rock Stockpile Site (3B-10 @ River Mile 10)	12	36	26	61
Total All Stations	21	40	55	96

DISCUSSION

In the fields of engineering and mathematics, a given amount of change in an input parameter usually results in a very predictable, precise, and immediate response in an output function. For example, increasing the opening size of an outlet valve in a dam will result in a precise increase in the amount of water being released downstream. It would be wonderful if a trout population in a tailrace fishery (such as the Dolores River below McPhee Dam) would have a precise and instantaneous response to a change in the flow regime. Unfortunately, trout do not respond in an instantaneous manner to stress factors (short of a dry streambed or a drained lake) in the environment any more than humans or other animal species do. Rather, the impacts of habitat stress factors have a slow, cumulative effect over an undefined period of time. These effects are manifested in increased mortality rates among the most susceptible components of the trout population. Thus, it is only through long—term (years) biological monitoring programs that the responses of an organism or population to environmental stress (es) can be documented.

Fortunately, the CDOW began a systematic monitoring of the Dolores River (below McPhee Dam) trout population in 1987 that has continued up to the present time. This data base facilitated an in depth look at the long-term trout population response to the decreased stream flow and the concomitant decrease in trout habitat quality and quantity in the Dolores River in 1989 and 1990.

The dramatic decline in trout density and biomass began at river mile 10 (the most downstream monitoring point) after October 1988. Similarly, trout density and biomass at river mile 6 peaked in the fall of 1989 and has been on the decline since that time. Finally, density and biomass remained relatively stable at river mile 0.5 between September 1989 and October 1990. However, a dramatic decline occurred at this station between October 1990 and April 1991, in spite of the fact that no icing occurred during the winter of 1990/1991 at this site!

Naysayer's and skeptics might look to other possible causes (for the decline), perhaps suggesting the CDOW no longer stocks fingerling trout in the Dolores River corridor between P4cPhee Dam and Bradfield Bridge. Nothing could be further from the truth as the information in Table 10

shows. Near record numbers of fingerling rainbow and cutthroat trout have been stocked in 1988, 1989, and 1990. Stocking of brown trout fingerling was terminated after 1988 as the brown trout population is currently self-sustaining through natural reproduction (see Table 9). The age and growth information in Table I-i (Appendix) indicates that the average size of age 1+ rainbow, brown, and cutthroat trout is 20-25 cm. Similarly, the average size of all age 2+ trout is 32-40 cm. Age 3+ trout range in size from 35 - 48 cm. Thus, fingerling trout stocked in 1987 and 1988 would have all been in the 20 to 40 cm size range by October 1990.

Table 10. Summary of the fingerling trout stocking history for the Dolores River below McPhee Dam, 1984 - 1990.

Species	1984	1985	1986	1987	1988	1989	1990
Brown	5,000	10,000	16,000	6,100	3,000	-----	-----
Rainbow	10,000	10,000	-----	-----	5,000	29,000	14,300
Cutthroat	10,000	10,000	-----	10,000	15,000	15,000	15,000
Total Trout	25,000	30,000	16,100	16,100	23,000	44,000	29,300

However, the data in Table 7 indicates that the largest decreases in trout density and biomass in October 1990 and April 1991 (compared to the fall of 1988 and 1989) have occurred in the 20 - 40 cm size range. Losses of the larger (and older) trout have been less dramatic. Total trout density and biomass (for trout 2 20 cm & ~ 40 cm) at river mile 10 decreased 82% and 88%, respectively between October 1988 and April 1991. In contrast, at river mile 6, while total density and biomass for trout 2 40 cm) remained relatively stable between September 1989 and October 1990, total density and biomass (for trout 20 - 40 cm) decreased by 42% and 51%, respectively. Similarly, at river mile 0.5, total density and biomass (for trout 2 40 cm) increased by 81% and 105% between September 1989 and October 1990. At the same time total density and biomass (for 20 - 40 cm trout) decreased by 23% and 38%. respectively, at this site.

In all instances (for 20 — 40 cm trout) at all three sites the percent decrease in biomass was greater than that of density. This clearly indicates a greater decline occurred in trout body condition than in numbers (density) of trout; another indicator of severe stress during the summer of 1990. The question then becomes, “What are the mechanisms or impacts that apparently lead to excessive mortality of these intermediate (20 - 40 cm) size classes?” Most likely it is due to the compounding effects of two factors: (1) reduced habitat availability as a result of the low stream flows and warmer water temperatures during the summer of 1990, and (2) increased competition and agonistic encounters between larger older (and therefore more dominant) trout and the smaller less dominant (20 — 40 cm) trout.

Addressing the second factor first, several studies of stream—dwelling trout indicate that a hierarchical dominance or pecking order, exists among conspecifics (Bachman 1984) as well as between different species (Fausch and White 1981; Fausch and White 1986; Fausch 1988). Generally larger body size confers dominance with the dominant fish occupying and defending the most advantageous position within its home range (Bachman 1984; Fausch and White 1981). It seems natural to assume that agonistic encounters and other aggressive interactions would become more frequent during periods of severe habitat restriction.

Returning to the first hypothesis, i.e.. reduced habitat availability can be linked to reduced stream flow, several simple generalizations can be made. First of all, as discharge decreases the channel

becomes both shallower and narrower, thereby reducing the acreage of the channel under water. This is a decrease in habitat in the simplest sense.

However, the totality of habitat needs for any species is a complex and dynamic process. Trout "habitat" is made up of many things and factors that are in a constant state of flux over time. Food quantity and quality, water temperature, dissolved oxygen, cover, chemical parameters such as alkalinity, hardness, pH, and naseum, are just a few of the parameters that are important ingredients in what fishery biologists call "trout habitat". Many attempts have been made to model stream trout habitat. Indeed, Fausch et al (1988) reviewed 99 different models for quantifying trout standing crop from habitat variables. Many of these models have serious drawbacks.

However, the IFIM/PHABSIM (Instream Flow Incremental Methodology/ Physical Habitat Simulation) models (Milhous et al 1984) are increasingly recognized as one of the more reliable methodologies for quantifying stream flow and fish habitat (Reiser et al 1989). This modeling process has the capability of converting water depth and velocity measurements across a stream transect into physical habitat units used by trout. These habitat units are called weighted usable area (WUA). Nehring and Shuler(1991) used the IFIM/PHABSIM models to quantify the amount of physical habitat (WUA) in nine study sites on the Rio Grande River near South Fork, Colorado. They found statistically significant correlations between adult brown trout (2 29 cm) and adult brown trout habitat (WUA) over the nine study sites for a two year period (1989-1990). Earlier studies (Nehring and Anderson 1984; Nehring and Miller 1987; Nehring 1988) conclusively demonstrated a very strong link between trout fry habitat availability and subsequent trout recruitment and year class strength. Rainbow and brown trout fry habitat availability was directly linked to stream discharge.

The IFIM/PHABSIM stream habitat modeling process has been used on the Dolores River below McPhee Dam to elucidate the stream discharge/trout habitat relationship. Three different management agencies (U.S. Bureau of Reclamation, Bureau of Land Management, and the CDOW) have modeled various sections of the Dolores River below McPhee Dam. The CDOW and USBR sites overlap and are located approximately 0.5 miles downstream of McPhee Dam. These study sites are quite representative of the Dolores River corridor between the dam and Bradfield Bridge. The BLM study site is located farther downstream in the incised canyon between Bradfield Bridge and Slick Rock, Colorado, and is therefore less representative of the section of river addressed in this report.

Habitat suitability curves for depth (feet), velocity (ft/s), and substrate size (Brusven 1977) for the various life stages of brown and rainbow trout were used to generate the habitat (WUA)/discharge relationship given in Table 11. The data in Table ii contains the results of the IFIM/PHABSIM analysis for the CDOW site. This data was collected in July 1986 and the analysis was completed in March 1987. The cross sectional profiles in the study area were installed prior to the time when stream habitat improvement had been completed. The USBR data set was collected during the summer of 1990 and the data analysis has not yet been completed. Therefore, no comparisons of the two data sets are possible at the present time.

In assessing any WUA/discharge relationship several notes of caution must be kept in mind. First, the actual habitat units (WUA) at any discharge are not what is most important in the evaluation. Rather, it is the shape of the WUA/discharge curve and the relative increase or decrease in habitat value with a change in discharge. Minor changes in the habitat suitability curves (for depth, velocity, and substrate) can potentially produce large changes in estimated habitat (WUA) at any given flow. Finally, cover (instream structure) and water temperature suitability criteria curves were not included in the analysis for the Dolores River. Elevated water temperatures (as occurred during the late spring-early summer of 1990) and lack of cover (which would become progressively more critical but less available to trout as flow decreases) would have a compounding synergistic effect on the Dolores River trout population(s).

Table 11. Summary of adult brown and rainbow trout habitat (WUA) vs discharge (cfs) per 1000 feet of channel for the Dolores River.

Flow (cfs)	Adult Brown WUA	<u>Adult Rainbow WUA</u>
20	18962	4942
40	25332	9808
50	36998	19884
75	41544	28764
100	42331	36189
150	41772	48334
200	39378	54286
250	36555	54247
300	32927	51301
500	19713	29881
750	11169	12972
1000	9577	9794

Returning to the WUA/discharge data in Table 11, adult brown trout WUA reaches its maximum value at 100 cfs, but remains very near the maximum value between 75 and 200 cfs. Similarly, adult rainbow trout WUA reaches its maximum value at 200 cfs and remains near maximum value between 150 and 250 cfs.

In contrast, at 50 cfs, adult brown trout WUA dropped 13% from its maximum at 100 cfs, This small change may not have been critical for adult brown trout in the Dolores River below McPhee Dam in 1990. However, at 40 cfs adult brown trout WUA is reduced 40% from the maximum (at 100 cfs). At 20 cfs adult brown trout WUA was reduced by 55% at a time when daytime air temperatures were ranging up to 90 — 100 F.

The impacts of flow reductions to 20 cfs on adult rainbow trout WUA (during the spring of 1990) were far more dramatic. Rainbow trout prefer faster water than do brown trout; thus, adult rainbow trout WUA reaches its maximum levels at 200 cfs. At a discharge of 50 cfs, adult rainbow trout WUA is reduced 63% from the maximum value at 200 cfs. At 40 cfs it is reduced 82% and 20 cfs it is reduced 91%! Again, this does not account for the compounding effects of elevated water temperatures and decreasing amounts of cover under the reduced flow regime.

Given these levels of adult trout habitat losses from March through June 1990, it is interesting to note the percent decrease in 20-40 cm (8 - 16 inches) brown and rainbow trout biomass in April 1991, compared to the peak biomass levels observed during the fall of 1988 and 1989 at the various stations (see Table 7). Adult rainbow trout biomass (in the 20-40 cm size group) decreased by 76%, 89%, and 83%, respectively, at river mile(s) 0.5, 6, and 10. Similarly, adult brown trout biomass (for the 20 — 40 cm size classes) decreased from the maximum values (in the fall of 1988 and 1989) by 84%, 84%, and 89%, respectively, at river mile(s) 0.5, 6, and 10. Finally, all Snake River cutthroat trout (in the 20 - 40 cm size range) were absent at all three study stations both in the fall of 1990 and April 1991.

It may have taken 6 months to a year for these devastating losses to occur (after the flow reduction to 20 cfs), but it would be difficult to find a more graphic response of a trout population to a severe habitat reduction than what has occurred on the Dolores River below McPhee Dam.

Water temperatures into the upper 70 F range during the summer of 1990 only served to exacerbate the habitat depletions as a result of low water levels. Coutant (1977) indicates that the preferred water temperatures for adult brown and rainbow trout are in the 12 - 13 C (53.6 — 55.4 F) range while they avoid water temperatures of 19 — 20 C (66.2 - 68 F). Elliott (1975a,b), conducting studies on the effect of water temperature on the growth rate of brown trout, found that the maximum rate of growth occurred in the temperature range of 12.8 — 13.6 C (55 — 56.5 F). He also found that for brown trout fed reduced rations the rate of weight loss grew increasingly worse the higher the water temperature went above 13 C. At water temperatures of 19.5 C the growth rate was zero even though trout were maintained on maximum ration size, i.e., fed all the food they would consume.

Minimum Flow Regimes of the Dolores River

A simple comparison of the minimum flow regime for the Dolores River to other tailrace trout fisheries around the state of Colorado gives an easy indication of just how bad the flow regime has been in the Dolores River during the summer of 1990. These comparisons are given in Table 12. There are only two reservoirs on major drainages in Colorado with minimum flow levels released for downstream trout fishery needs that even begin to approach the minimum flows on the Dolores River below McPhee Reservoir. They are Dillon and Cheesman reservoirs. both operated by the Denver Water Department.

Table 12. Comparison of minimum flow and average discharge data for major drainage trout streams controlled by reservoirs in Colorado.

River and gage location (Dam)	AF/Yr ¹ x1000	Av. Q ² cfs	Years	Discharge Data				
				1990	No. Da.	10 Yr/30 da	1990	
				min Q cfs	Duration	Minimum Q cfs % Ave	MinQ % Ave	
Dolores River (Below McPhee)	315.2	435	79	20	101	20 4.6	4.6	
Colorado R. @ Hot Sulphur Spgs (Below Granby)	173.9	240	37	64.3	30	55.2 23.0	26.7	
Fryingpan R. (below Ruedi)	132.6	183	23	80	180	34.3 18.7	43.7	
Gunnison R. below Tunnel (Crystal Dam)	994	1372	87	294	30	226 16.5	21.4	
Blue River (below Dillon)	152.1	210	27	52	130	21.5 10.2	24.3	
South Platte R. (below Cheesman)	123.2	170	66	35.5	30	23.8 14.0	17.6	
Taylor River (below Taylor)	141.3	195	52	97	60	50.3 25.8	49.7	

1= Acre-feet/year x 1000
2=Q means discharge in ft³/s.

None of the major trout streams listed in Table 12, in the last 10 years, have seen 30 day minimum flow levels that equated to less than 10% of the average discharge. The South Platte River below Cheesman Reservoir was the only major trout stream to have a 30-day minimum flow less than 20% of the average discharge during the 1990 water year.

In contrast, the 20 cfs minimum flow (which occurred for 101 days) in the Dolores River below McPhee Reservoir during 1990, equates to 4.6% of the average discharge of the Dolores River into McPhee Reservoir for 79 years of record. Even the "wet year" flow of 78 cfs only equates to 17.9% of the average discharge as a minimum flow release for downstream fisheries.

SUMMARY

Drought in southwestern Colorado during the summers of 1988, 1989, and 1990 resulted in severe streamflow reductions in the Dolores River below McPhee Reservoir beginning in March 1990 and extending through April 1991. The combined effects of reduced flows, unusually hot spring air temperatures, elevated water temperatures, dramatically reduced levels of adult trout habitat, and silt accumulation manifested themselves in dramatic declines in trout density and biomass in the Dolores River corridor between McPhee Dam and the Bradfield Bridge (12 miles downstream).

Total trout density (n/ha) and biomass (kg/ha) began declining in 1989 at the most downstream electrofishing station (10 miles below McPhee Dam). Density and biomass at this station in April 1991 had declined 59% and 71% respectively, from the fall of 1988. Density and biomass at river mile 6 (in April 1991) had declined 55% and 35% respectively, compared to the fall of

1989. Finally, density and biomass at river mile 0.5 decreased 51% and 53% between October 1990 and April 1991.

The largest losses in the trout population were among the intermediate size and age groups. Total trout density and biomass in the 20-40 cm (8—16 inches) size range declined by 82% and 88%, respectively, at river mile 10 between October 1988 and April 1991. At river mile 6, total trout density and biomass (for 20-40 cm trout) decreased by 42% and 51% between September 1989 and October 1990. And at river mile 0.5, total trout density and biomass for the 20—40 size group decreased by 23% and 38%, respectively, for the same time period.

Severe habitat reductions, as shown with the IFIM/PHABSIM models, most likely resulted in severe stress and competition for food and living space among the two year old and older trout in the population. Agonistic encounters between the very large trout (2 40 cm or 16 inches) and the smaller (and therefore less dominant) 20 to 40 cm (8 — 16 inches) trout would result in the smaller trout being pushed out into peripheral less desirable habitats where they would be more vulnerable to disease, predation, heat stress, weight loss, and ultimately death.

Comparison of the minimum flow levels in the Dolores River below McPhee Dam with the minimum flow regimes of several other tailrace trout streams across Colorado reveals that this river suffers far worse flow depletions for much longer periods of time than any other major stream trout fishery in Colorado. The 20 cfs flow in the Dolores River (for 101 days in 1990) equates to 4.6% of the average annual discharge. No other major trout stream in Colorado has had the flow reduced to less than 10% of its respective average discharge in the last 10 years.

Losses of trout in the Dolores River below McPhee Dam between October 1990 and April 1991 can not be attributed in any measurable or detectible way to the effects of icing during the winter of 1990—1991.

The largest significant decreases in trout density and biomass for this time period occurred at river mile 0.5, the only station that received no icing at any time! The minimum water temperature at this station during the winter of 1990/1991 was never less than 37.5 F. In contrast, the other two stations (at river miles 6 and 10) were totally frozen over from late-December 1990 through mid—February 1991, yet over winter losses in density and biomass were relatively minor at these two stations.

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Table I-1. Dolores River age, growth, and back-calculated length data, Fall 1988 - Fall 1990.

Year	Age	N	L_{ci}	S.E.	L_1	S.E.	L_2	S.E.	L_3	S.E.	L_4	S.E.	L_5	S.E.
Brown Trout - Fall 1990														
1990	0+	5	7.6	0.40										
1989	1+	2	17.0	1.00	11.9	1.12								
1988	4+	1	47.0	-----	15.7	-----	32.4	-----	37.7	-----	42.2	-----		
Rainbow Trout - Fall 1990														
1990	0+	2	12.0	-----										
1989	1+	4	15.3	-----	10.0	0.14								
1987	3+	2	45.0	2.00	14.9	1.11	35.3	2.61	42.9	2.32				
1986	4+	1	47.0	-----	8.8	-----	18.7	-----	34.3	-----	39.9	-----		
Brown Trout - Fall 1989														
1989	0+	3	10.7	0.33										
1988	1+	8	22.4	1.74	12.8	0.43								
1987	2+	8	32.9	1.06	14.0	1.00	26.8	1.10						
1986	3+	8	37.8	1.10	11.7	0.55	22.9	2.16	33.9	1.44				
1985	4+	5	42.0	1.76	11.0	0.96	18.9	1.23	32.9	1.49	38.8	1.85		
1984	5+	2	46.5	1.50	11.2	1.90	18.1	1.66	37.7	0.75	41.1	1.94	44.0	1.54
Rainbow Trout - Fall 1989														
1988	1+	10	24.2	0.85	16.2	0.30								
1987	2+	9	37.0	1.24	17.9	0.87	29.7	1.42						
1986	3+	11	40.5	0.87	17.5	0.84	28.9	1.64	37.3	0.77				
1985	4+	2	45.5	3.50	17.2	1.50	27.8	1.51	36.5	2.98	42.4	31.6		
Snake River Cutthroat Trout - Fall 1989														
1989	0+	2	17.5	0.50										
1988	1+	4	23.5	0.29	12.1	1.16								
1987	2+	7	40.7	1.94	15.8	0.89	32.0	1.91						
1986	3+	2	48.0	2.0	15.8	2.11	37.3	0.83	44.0	1.14				

Table I-1.(continued) Dolores River age, growth, and back-calculated length data, Fall 1988 - Fall 1990.

Year	Age	N	L_{t1}	S.E.	L_1	S.E.	L_2	S.E.	L_3	S.E.	L_4	S.E.	L_5	S.E.
Brown Trout - Fall 1988														
1988	0+	7	10.3	0.18										
1987	1+	38	21.4	0.50	12.1	0.41								
1986	2+	16	32.6	1.16	13.1	0.89	24.7	1.36						
1985	3+	15	34.5	1.01	9.97	0.51	18.5	0.75	29.5	1.17				
1984	4+	7	42.6	1.40	11.6	1.57	23.4	3.63	31.5	2.76	39.3	1.66		
Rainbow Trout - Fall 1988														
1988	0+	31	13.1	0.51										
1987	1+	12	21.7	0.64	13.8	0.64								
1986	2+	3	36.3	0.88	13.9	1.81	27.8	0.56						
1985	3+	21	39.3	0.78	12.7	0.40	22.4	1.21	35.3	0.63				
1984	4+	6	39.8	1.40	11.5	0.93	18.2	1.89	30.0	2.31	35.9	1.49		
1983	5+	1	49.0	----	15.1	----	29.7	----	37.6	----	42.7	----	46.5	----

Table I-2. Visual implant (VI) tagging data for the Dolores River below McPhee Dam, April 4-5, 1991.

Station 3B-10			Station 3B-15			Station 3B-20		
Tag Code	Species	Age	Tag Code	Species	Age	Tag Code	Species	Age
TA1	R47		TA3	B43		TB6	R47	
TA2	R43		TA5	B43		TB7	R47	
ZA0	B31		TA6	B50		TB8	R40	
ZA1	B43		TA7	B43		TB9	R43	
ZA2	B20		TA8	C41		TC0	B42	
ZA3	B21		TB0	C47		TC1	R43	
ZA4	B21		TB1	C42		TC2	R44	
ZA5	B25		TB2	B44		TC3	C44	
ZA6	B27		TB3	R45		TC4	R42	
ZA7	R21		TB4	R45		ZD3	R23	
ZA8	B20		ZC0	B23		ZD4	R25	
ZA9	R21		ZC1	B38		ZD5	B26	
ZB0	C19		ZC2	R29		ZD6	B23	
ZB1	B18		ZC3	B30		ZD7	R22	
ZB2	R24		ZC4	B25				
ZB3	B20		ZC5	B23				
ZB4	B19		ZC6	R36				
ZB5	B22		ZC7	B24				
ZB6	B33		ZC8	B23				
ZB7	B22		ZC9	B24				
ZB8	B20		ZD0	B22				

Table I-3. Summary of some physical habitat characteristics used by brown, rainbow, and cutthroat trout in the Dolores River below McPhee Dam, January-February, 1991.

Size Category			Depth (feet)		Nose Velocity ¹		Mean Column Vel. ¹		
No	In.	cm.	N	Mean	Range	Mean	Range	Mean	Range
Brown Trout									
0	≤4	≤10	4	1.9	1.2-2.9	0	-----	0	-----
1	4-8	10-20	11	1.94	0.6-2.6	0.23	0 - 0.47	0.42	0 - 0.91
2	8-12	20-30	14	2.30	0.7-3.4	0.38	0 - 0.84	0.56	0.12-1.06
3	≥12	≥31	8	2.65	1.6-3.7	0.30	0 - 0.47	0.46	0.19-0.70
Rainbow Trout									
0	≤4	≤10	16	1.29	0.6-1.9	0.34	0 - 0.99	0.53	0 - 1.64
1	4-8	10-20	18	1.68	0.7-2.7	0.16	0 - 0.77	0.34	0 - 1.13
2	8-12	20-30	8	2.43	0.9-4.0	0.31	0 - 0.84	0.61	0 - 1.38
3	≥12	≥31	14	2.07	1.0-3.0	0.43	0 - 0.99	0.59	0 - 1.31
Snake River Cutthroat Trout									
3	≥12	≥31	4	3.40	2.0-4.2	0.33	0.26-0.4	0.56	0.47-0.66

¹ Velocity is in feet/second