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Intermountain Journal of Sciences

The Intermountain Journal of Sciences is a regional peer-reviewed journal that encourages scientists, educators and students to submit their research, management applications or view-points concerning the sciences applicable to the intermountain region. Original manuscripts dealing with biological, environmental engineering, mathematical, molecular-cellular, pharmaceutical, physical and social sciences are welcome.

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Conservation of Native Inland Fishes of the West

Guest Editors

Carter G. Kruse
Bradley B. Shepard

Underwriters

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Three hard copies of the submitted manuscript, with copies of the "Guidelines and checklist for IJS referees" attached are forwarded to the appropriate Associate Editor. The Associate Editor retains one coy of the manuscript and guidelines for his/her review, and submits a similar package to each of two other reviewers. A minimum of two reviewers, including the Associate Editor, is required for each manuscript. The two other reviewers are instructed to return the manuscript and their comments to the Associate Editor, who completes and returns to the EIC a blue "Cover Form" and all manuscripts and reviewer comments plus a recommendation for publication, with or without revisions, or rejection of the manuscript. This initial review process is limited to 30 days.

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Each manuscript that is rejected for publication is returned by the EIC to the corresponding author along with the reasons for rejection. The author is also advised that the manuscript may be resubmitted, provided all major criticisms and comments have been addressed in the new manuscript. The new manuscript may be returned to the initial review process if deemed appropriate by the EIC. If the manuscript is rejected a second time by either the EIC or the Associate Editor and reviewers, no further consideration will be given for publication of the manuscript in IJS. The corresponding author will be notified of this decision.

Reviewer Anonymity

The identity of all reviewers shall remain anonymous to the authors, called a blind review process. All criticisms or comments by authors shall be directed to the EIC; they may be referred to the ME or the Editorial Board by the EIC for resolution.

Manuscripts Submitted by Editors

Each manuscript submitted by an Associate Editor shall be reviewed by the EIC and a minimum of two other reviewers with expertise in the subject being addressed. Each manuscript submitted by the EIC shall be forwarded with the necessary review materials to the Chairman of the Editorial Board of IJS, who will serve as the EIC for that manuscript.

Abstracts

Only abstracts from the annual meetings of the sponsoring organizations will be published in IJS. Other submissions of abstracts shall be considered on a case-by-case basis by the Editorial Board. Sponsoring organizations shall collect abstracts, review them for subject accuracy, key or scan them onto a 3.5" diskette, and submit the diskette and hard copy of each abstract to the EIC on or before November 1. Each abstract shall be reviewed by the EIC to assure proper grammar, compliance with IJS "Guidelines for Abstracts Only" and for assignment to the appropriate discipline section. All abstracts will be published in the December issue only.

Commentary

Submissions concerning management applications or viewpoints concerning current scientific or social issues of interest to the Intermountain region will be considered for publication in the "Commentary" Section. This section will feature concise, well-written manuscripts limited to 1,500 words. Commentaries will be limited to one per issue.

Submissions will be peer reviewed and page charges will be calculated at the same rate as for regular articles.

Literature Cited

Dusek, Gary L. 1995. Guidelines for manuscripts submitted to the *Intermountain Journal of Sciences*. Int. J. Sci. 1(1):61-70.

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Fish illustrations courtesy of Joseph Tomellari
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Synopsis of the Symposium

Practical Approaches for Conserving Native Inland Fishes of the West

Bradley B. Shepard, Montana Fish, Wildlife and Parks and Montana Cooperative Fishery Research Unit, Bozeman, MT

The Montana Chapter and Western Division of the American Fisheries Society (AFS) hosted a symposium entitled **"Practical Approaches for Conserving Native Inland Fishes of the West"** 6-8 June 2001 at the University of Montana in Missoula. The purpose of this symposium was to bring together fishery professionals from throughout the western United States and Canada to share their ideas and experiences about conservation of native fish in the region. As the keynote speaker, Dr. Bruce Rieman, noted conservation management of fishes is a discipline in transition with threats to native fishes increasing while management goals and our understanding of these systems are changing. Here, I present a brief review of that symposium and introduce five papers appearing in this issue that resulted from the symposium. The National Fish and Wildlife Foundation graciously paid publication costs for this issue of the *Intermountain Journal of Sciences* in order to disseminate this information. Extended abstracts covering the entire symposium (Shepard 2001) can be found on the Montana Chapter AFS web site at (http://www.fisheries.org/AFSmontana/) under "Special Features," **Abstracts from the Practical Approaches to Conserving Inland Fish Species Symposium**.

The symposium included 49 oral presentations and 6 poster presentations and covered a broad range of topics relevant to native fish: conservation agreements and the value of native fish, conserving fish in large river systems, impacts of nonnative fish on native fish, methods for removal of nonnative fish, habitat use and restoration, genetic issues, miscellaneous contributed papers, and a case study.

Conservation of native fish will require that management be more consistent with natural processes, but also will need to include some type of "reserves" (Rieman 2001). Conservation must be an adaptive process because we are continually learning new things about both population and ecosystem processes and changing our management goals and objectives (Rieman 2001). Native fish are valuable for anthropocentric, biocentric, legal, and spiritual reasons and one should be aware that different individuals view native fish differently based on their own personal value system (Wiltshire 2001).

The U.S. Endangered Species Act (ESA) is an overarching law that regulates conservation of native fish and criteria used for evaluating conservation efforts under this law are unclear (Gloman 2001, Lentsch and Toline 2001, Phillips 2001, Watson et al. 2001). Conservation agreements, for both state and private entities, are one legal avenue for potentially meeting legal requirements of ESA (Kruse 2001); however, to be effective these agreements must demonstrate that sound science is being used, all appropriate people and agencies are collaborators, monitoring is included, the program is funded, there is a track record of success, and individual conservation agreements are part of a larger plan to ensure a species is conserved throughout a large geographic area (Phillips 2001). Conservation efforts will probably be more effective if they begin before a species declines to a level where extinction is likely (Gloman 2001, Lentsch and Toline 2001, Phillips 2001).

Conservation of native fish in large rivers presents a daunting challenge, especially since most river systems have been altered by large impoundments. Fertilization has been used to restore nutrient inputs and cycling in large systems that have large impoundments that act as nutrient sinks (Ashley et al. 2001, Marotz et al. 2001). Flow and water temperature management, conservation aquaculture, fish passage, and harvest restrictions were also either used or proposed to conserve native fishes in large rivers (Gardner 2001, Marotz et al. 2001, McMahon et al. 2001, Muth et al. 2001, Paragamian et al. 2001). Flow variation, particularly peak flow events, was linked to native fish community structure in a Southwestern river (Rinne 2001).

Nonnative fish threaten many native fish populations. In the past the two chemicals rotenone and antimycin-A have been used to remove nonnative fish, but their continued use depends upon legal registration as piscicides and upon public acceptance that their use is warranted and safe (Brawer 2001, Brooks and Propst 2001, Clancey 2001, Finlayson 2001, Hepworth et al. 2001, Joscelyn 2001, Moore et al. 2001, Schnick 2001, Sexauer 2001, Stevens 2001, Wheelis 2001). Holcomb (2001) presented techniques and tips for effectively communicating potential risk of these chemicals to the public. Nonnative fish have been removed using piscicides (Buktenica et al. 2001, Brooks and Propst 2001, Clancey 2001, Holden et al. 2001, Sexauer 2001, Stevens 2001), electrofishing (Buktenica et al. 2001, Moore et al. 2001, Shepard and Spoon 2001a, 2001b), angling (Stelfox et al. 2001), seines (Holden et al. 2001), and pheromonal attraction to hoop nets (Young 2001). The effectiveness of these removal techniques and the response of native fish to these removals were variable.

Temperature was shown to be an important viable influencing habitats selected by native fish and may potentially mediate interactions between native and nonnative fish (Ebersole et al. 2001, Haas 2001, Muth et al. 2001, Sloat et al. 2001). Life history variability, ontogenetic habitat shifts, and connectivity between habitats are all probably important for maintaining bull trout (Carnefix et al. 2001) and coastal cutthroat trout (Hendricks and Gresswell 2001) populations. Diel habitat shifts were observed for juvenile bull trout in a large river system (Muhlfeld et al. 2001). Restoration efforts to improve and connect habitats in the Blackfoot River appeared to increase abundances of both native bull and westslope cutthroat trout (Peters et al. 2001, Schmetterling 2001). Connectivity or isolation of a native fish population can both be used as management tools depending upon threats to a particular population (Hepworth et al. 2001, Knotek et al. 2001, Peters et al. 2001, Schemetterling 2001, Sexauer 2001, Shepard and Spoon 2001a, 2001b). Habitat restoration was shown to enhance an allopatric native westslope cutthroat trout population (Oswald 2001). Brown et al. (2001) and Miller and Skidmore (2001) descibed philosophies and approaches for restoring stream habitats.

Genetic analyses can provide important information to infer how much isolation individual populations may have experienced over time (Leary 2001, Spruell 2001). This information could be used to determine whether a particular conservation effort should focus on connectivity or isolation among individual stocks of fish. Spruell (2001) suggested as most reasonable a hierarchical conservation approach that uses genetic, demographic, ecological, and environmental variability. Spruell et al. (2001) also suggested a hypothetical application of this type of approach for bull trout in the lower Clark Fork River drainage in Montana and Idaho.

The following five papers provide more detailed information on various aspects of native fish conservation. Dale Hepworth and his co-authors review 25 years of native trout conservation in southern Utah. Wade Fredenberg reports on the displacement of native bull trout by introduced lake trout in Glacier National Park. Matt Sloat and his co-authors relate the presence of dispersal barriers and water temperature to the distribution of native westslope cutthroat trout in the Madison River drainage of

Montana. Vaughn Paragamian and his co-authors explain how they developed an international strategy for recovering burbot in the Kootenai River/Kooentay Lake drainage of Idaho and British Columbia. Finally, my co-authors and I describe a project to remove nonnative brook trout and restore stream habitat degraded by past placer mining to conserve a small population of native westslope cutthroat trout in a tributary to the Missouri River in Montana.

Thanks to the following organizations and individuals who provided major funding for the symposium: National Fish and Wildlife Foundation (P. McClelland and B. Ocepek); USDI Fish and Wildlife Service, Denver Region (M. Stempel); and Plum Creek Timber Company (G. Watson and L. Hicks). Contributors also included the: Kootenai Tribe of Idaho (S. Ireland); Turner Enterprises Incorporated (M. Phillips and C. Kruse); USDI Bureau of Reclamation (S. Camp and J. Jacobs); Federation of Fly Fishers (B. Wiltshire and C. Marosek); PPL Montana (B. Mabbott); Wyoming Game and Fish (D. Miller); Idaho Fish and Game (V. Moore and B. Hutchinson); Northern Region of U.S. Forest Service (P. Van Eimeren, B. Riggers, M. Jakober, W. McClure, L. Walch, S. Phillips, and J. Brammer); Montana Fish, Wildlife and Parks (C. Hunter); and the USDI Bureau of Land Management (J. Silvey and M. Whisler). Sponsors of events and scholarships included: Urbani and Associates (J. Urbani); Montana Power Company (S. Milodragovich); Drake and Associates (B. Drake); Confederated Salish and Kootenai Tribes (L. Everts); and Montana Council of Trout Unlimited (B. Farling). I want to acknowledge the great job the University of Montana's Conference Services did in hosting the symposium and producing the abstracts for printing and web distribution. I would especially like to thank all the Montana Chapter AFS members who helped put on the symposium and the Western Division AFS officers who supported us in this effort.

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A Review Of A Quarter Century Of Native Trout Conservation In Southern Utah

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Abtract

Status of native cutthroat trout first became a management issue in southern Utah in the 1970s after the Endangered Species Act was passed and several remnant populations of native trout were identified. Initial restoration efforts began in 1977 when individuals from a remnant population of Bonneville cutthroat trout (*Oncorhynchus clarki utah*) were transplanted to a stream that had been treated with rotenone to remove nonnative trout. Restoration efforts became a routine part of Utah's fishery program in 1980 when the state incorporated Federal Aid in Fish Restoration funding into its program, which formally included native trout. Here, we evaluate the native trout restoration program by reviewing the progress made and problems encountered during the past quarter century. Evaluations were categorized by topic: (1) implications of changing genetic identification techniques; (2) success of treating streams and lakes with rotenone; (3) sources of native trout for re-introductions; (4) use of migration barriers to isolate native from nonnative trout; (5) practical considerations in restoration of metapopulations; and (6) socio-political issues. Project delays, setbacks, and failures have occurred over time, but overall accomplishments have been positive. Consistent progress resulted from making native trout restoration a formal part of annual work plans. Stream habitat known to contain native trout has increased over 15 times since 1977. Wild brood stocks were developed from local sources of both Bonneville and Colorado River (*O. c. pleuriticus*) cutthroat trout. Plans are in progress to develop additional stream and lentic populations of native cutthroat trout, and incorporate native trout into overall sport fishery management plans.

Key words: conservation, Cutthroat trout, failure, native, quarter century, restoration, review, southern Utah, success

Introduction

Declines in abundance of native trout in Utah are attributed to factors which have been widely acknowledged to cause declines of cutthroat trout throughout western North America. These include hybridization and displacement of native trout from introductions of nonnative trouts, loss of habitat, and, to a lesser extent, exploitation from angling. Conservation and restoration of native trout in southern Utah became part of fishery management objectives for the Utah Division of Wildlife Resources (UDWR) in the 1970s after passage of the Endangered Species Act (ESA), as amended in 1973, and discovery of several populations of native Bonneville cutthroat trout (*Oncorhynchus clarki utah*; Behnke 1976). Native trout programs were well established in the 1980s once Federal Aid in Fish Restoration funding (Dingell-Johnson Act) was institutionalized in regional fishery management programs that specified native trout projects and required formal annual reports. The primary management objective was to reduce threats that might lead to federal listing of native trout under the ESA. Southern Utah conservation projects were conducted for Bonneville cutthroat trout in the Sevier River drainage and part of the Virgin River drainage and for Colorado River cutthroat trout (*O.c. pleuriticus*) in the Escalante and

Figure 1. General location of the Sevier, Virgin, Fremont, and Escalante, river basins in the Southern Geographical Management Unit, Southern Utah, showing major rivers and tributaries.

Fremont river drainages (Fig.1). Most early work included identifying remnant populations of native trout and restoring and replicating these populations in historically occupied areas. Surveys to identify new populations and restoration projects were conducted concurrently because most of the sparse aquatic habitat in southern Utah had been previously surveyed and general distributions of native and nonnative trouts were known prior to initiation of projects targeted specifically at native trout.

Compared to earlier work, restoration efforts in the 1990s included more complex projects. Some projects included reservoirs and lakes that were connected to wild trout streams. Wild brood stocks of native trout were developed in two reservoirs that increased flexibility in management and allowed important sport fishery programs to be incorporated into restoration plans.

Our objective was to review conservation projects conducted in southern Utah since 1977 and categorize success, failures, and problems associated with specific management actions. We present a 25-year history of conservation efforts with discussion of important changes that occurred over time, including biological and social implications. Although we restricted

Table 1. Genetic tests conducted on remnant populations of native Bonneville and Colorado River cutthroat trout in southern Utah over time, and nature of barriers protecting genetic integrity. Genetic analytical techniques include meristics (M), allozymes (A), mitochondrial DNA (mD), and nuclear DNA (nD). Genetic test results include core population with <1 percent introgression (P), conservation population with <10 percent introgression (C) and sport fish population with >10 percent introgression (S).[1] Types of barriers include naturally isolated (N) due to barrier waterfalls or de-watered stream sections, planned artificial barrier constructed specifically to protect native trout (P), and unplanned artificial barrier (U) constructed as an unrelated water development project which coincidentally protected native trout.

Drainage / stream	Genetic tests, year and (method-results)	Stream length (km)	Mean stream width (m)	Barrier
	Bonneville cutthroat trout			
Sevier River				
Birch Creek-A	1973(M-P), 1976(A-P), 1990(A-P), 1990(mD-P)	5.5	1.19	N, P
N. Fk. North Creek	1981(M-P), 1981(A-P), 1990(A-P), 2001(nD-C,S)[2]	3.2	2.59	N, P
Deep Creek	1976(M-S)[3], 1981(A-P), 1995(mD-P), 2001(M-P)	9.7	1.86	N
Ranch Creek	1995(M-P), 1995(mD-P)	6.4	1.36	N
Virgin River				
Water Canyon	1976(M-P), 1976(A-P), 1987(M-P), 1987(A-P)	1.2	1.30	N
Reservoir Canyon	1976(M-P), 1987(M-P), 1987(A-P), 1993(mD-P)	3.2	2.35	N
	Colorado River cutthroat trout			
Escalante River				
E. Boulder Creek	1990(M-P), 1990(A-P), 1990(mD-P)	5.6	4.9	N
W. Boulder Creek	1993(mD-P) 2000(M-P)	3.2	2.6	U
W. Pine Creek	1997(M-P), 1997(mD-P), 1997(nD-P)	0.4	3.2	N
White Creek	1998(M-P), 2000(M-P) 2000(mD-P), 2000(nD-P)	1.8	2.0	N
Water Canyon	1997(mD-P), 1997(nD-P)	0.7	1.2	N

[1] Before 1996 introgression was not quantified, but classified as P (core population that was essentially pure) or S (sport fish population that was introgressed).
[2] Fish tested in 2001 from the headwaters of the North Fork North Creek tested 6% introgressed (conservation population), while fish directly above the migration barrier tested 15% introgressed (sport fish population).
[3] The first samples taken from Deep Creek were confused with samples from another stream.

this review to southern Utah and adapted much of what took place to local circumstances, many of the successes and failures might have implications for conservation of native fishes in other areas.

Methods

We reviewed data from all known populations of native trout in southern Utah, including remnant (naturally occurring) and restored populations, and ongoing restoration efforts currently in progress (Tables 1 and 2). During the late 1970s, 1980s, and early 1990s, surveys were conducted sporadically as needed to evaluate management actions with data compiled as UDWR file reports. During 1994-1995 upstream-downstream range and abundance of Bonneville cutthroat trout was determined for all known populations (Hepworth et al. 1997b). Similar surveys were conducted for Colorado River cutthroat trout in 1997 and 1998 (Hepworth et al. 2001). Surveys were repeated for most Bonneville cutthroat trout populations again during 2001 and spring of 2002 (data

Table 2. A summary of the restoration projects completed, planned, or in progress for Bonneville and Colorado River cutthroat trout in southern Utah. Types of barriers include natural barrier waterfall or de-watered section of stream (N), constructed single point barrier with nonnative trout immediately downstream (S), constructed barrier with additional obstacles(O) such as a de-watered stream channel preventing trout from occupying the stream below the barrier year-round, constructed multiple barriers (M) to create more than a single point obstacle, and unplanned artificial barrier (U) that was constructed for a primary purpose other than preventing fish passage. Status classifications include self-sustaining populations (S) established since the original restoration, conditionally or partially successful restoration (X) after supplemental actions taken to correct problems, unsuccessful (U) with problems resulting in project termination, and projects in progress (P) where native trout have not yet re-colonized areas being restored.

Drainage / stream / tributary, or reservoir	Year project initiated	Population origin	Stream length (km) or reservoir area (ha)	Mean stream width (m)	Barrier	Status
		Bonneville cutthroat trout				
Sevier River						
Sam Stowe Cr	1977, 1997	Birch Creek	4.8	1.43	O	X [1]
Pine Creek	1980	Water C., Reservoir C., Birch Creek	5.0	1.86	N	X [2]
Briggs Creek	1988	Birch Creek	1.4	1.0	N	S
Manning Res	1990	Pine Creek	23.1		O	X [3, 4]
Barney Reservoir	1993	Manning Reservoir	7.3`		O	X [4]
Threemile Creek	1994	Birch Creek	11.2	1.16	N, O	X [5]
Delong Creek	1994	Birch Creek	5.3	1.46	N	S
Indian Hollow	1994	Birch Creek	1.4	0.64	N	S
N. Fk. North Creek	1995, 1999	Remnant expanded	8.8	2.59	S	X [1]
Pole Creek	1995	N. Fk. North Creek	4.3		S	S
Manning Creek	1996	Manning Reservoir	17.2	2.90	N, O	S
Barney Creek	1996	Manning Reservoir	1.2	1.10	N, O	S
Vale Creek	1996	Manning Reservoir	1.6	1.70	N, O	S
E. Manning Cr	1996	Manning Reservoir	1.0	0.80	N,O	S
Sanford Creek	1999	Deep Creek	11.3		N, U	P
Sandy Creek	1999	Deep Creek	1.6		N, U	P
Birch Creek-B	2001	Manning Res	6.4		U	P
Tenmile Creek	2002	Deep Creek	9.7		O	P
Center Creek and Robs Reservoir	2002	Manning Res (planned)	12.0, 0.8		N	P
Virgin River						
Leap Creek	1986	Water Canyon	5.3	1.35	N	S
South Ash Creek	1986	Reservoir Canyon	7.1	2.35	N	S
Harmon Creek	1986	Reservoir Canyon	3.0	1.75	N	S
Mill Creek	1986	Reservoir Canyon	8.0	2.25	N	S
Leeds Creek	1989	Reservoir Canyon	11.3	2.71	N,U	S
Pig Creek	1989	Water Canyon	0.9	1.10	N	S
Spirit Creek	1988	Water Canyon	1.6	1.30	N	S
Horse Creek	1995	Spirit Creek	0.8	1.00	N	S
Spring Creek	1993		2.5			U [6]
		Colorado River cutthroat trout				
Escalante River						
Durfey Creek	1993	E. Boulder Creek	1.0		N	U [7]
Deer Creek	1994		2.5			U [6]

Table 2 (continued).

Drainage / stream / tributary, or reservoir	Year project initiated	Population origin	Stream length (km) or reservoir area (ha)	Mean stream width (m)	Barrier	Status
Dougherty Lake	1997	E. and W. Boulder creeks	1.5		N	X [3,4]
Tall Four Lake	2000	Dougherty Lake	0.3		N	X [4]
W. Boulder Creek	2001	Remnant expanded	9.6	2.6	M	P
Pine Creek	2001	Remnant expanded	8.0	2.7	M	P
White Creek	2001	Remnant expanded	2.1	2.0	M	P
Twitchell Creek and 2 Willow Bottom lakes	2001	Dougherty Lake	5.6, 4.6		N. O	P
Fremont River						
UM Creek	1996	Dougherty Lake	23.7	5.72	O	P
Left Fork	2000	Dougherty Lake	5.5	1.24	O	P
Right Fork	1996	West Boulder Creek	7.9	2.93	O	S
Sand Creek	1995	West Boulder Creek	4.8		N	X [8]
Forsyth Res	2000	Dougherty Lake (planned)	69.2		U	X [9]
Pine Creek and Pine Creek Res	2002	Dougherty Lake (planned)	15.1, 1.3		O	P

[1] Stream re-treated or partially retreated with rotenone after barrier failure.
[2] A second unplanned treatment was conducted.
[3] Wild brood stock.
[4] Supplemental stocking necessary at present in lake or reservoir.
[5] Nonative trout removed below barrier by electrofishing.
[6] Discontinued project due to socio-political ramifications.
[7] Stream habitat was not capable of supporting wild trout.
[8] Native trout re-introduced a second time after flash flood in lower stream.
[9] Impoundment on UM Creek temporarily drained, dam undergoing repairs.

available as file reports). Other surveys were conducted during the late 1990s and early 2000s on an as needed basis to make genetic evaluations, complete disease certifications, develop brood stocks, construct migration barriers, and evaluate other problems. Issues evaluated in terms of their implication to successful restoration for this study included (1) taxonomic and genetic analyses, (2) success of treating streams and lakes with rotenone, (3) sources of trout for brood stocks and re-introductions, (4) use of fish migration barriers, (5) practical considerations in the restoration of metapopulations, and (6) socio–political issues. Habitat evaluations and habitat improvement projects have been an important part of restoration efforts but were conducted by federal land management agencies and not included in this review.

Taxonomic evaluations used to identify remnant populations of native trout varied over time and were given varying levels of emphasis at different times as the state of genetic identification evolved. Throughout the entire study, cursory field observations of morphological characteristics were used to make putative identifications. Selected populations were further analyzed by submitting samples to university laboratories for meristic, allozyme, and mitochondrial and nuclear DNA analyses. We evaluated the consistency of genetic test results among methods over time and describe the implications of changing methodologies on restoration success. Similarly, treatment sites were evaluated by looking at successes and failures of rotenone application and subsequent restorations. We analyzed re-introductions on the basis of numbers of fish transplanted, time required to re-colonize renovated habitat, and sources of native trout with respect to utilizing wild trout and "nearest neighbors," i.e., closest available source of native trout, versus hatchery-produced fish from wild brood stocks. We did not evaluate fish migration barriers in terms of barrier dimensions but rather evaluated their role in terms of project success and circumstances under which barriers should be used. We evaluated metapopulation theory in the context of advantages, disadvantages, and practicality of re-establishing large, interconnected, and complex populations of native trout where they have been lost. Socio-political aspects of the evaluation were based on interplay among state and federal laws, inter-agency conservation agreements, agency policies and directives, and public interactions. The benefits and shortcomings of these laws, rules, and directives were considered relative to completing field projects.

We assessed the status of each restoration project using the above data and knowledge of each stream and lake. Restoration projects were classified as successful, conditionally or partially successful, unsuccessful, and in progress. Successful restoration projects were those where self-sustaining populations of native trout became established and have remained as such following completion of the originally scheduled restorations. Conditionally or partially successful restoration projects required supplemental actions to correct problems during years following completion of the initial project. Unsuccessful projects were those that were discontinued because of various problems. Projects in progress include those in which native trout have not yet fully re-colonized restored areas and become self-sustaining.

Discussion

Taxonomic and Genetic Analysis

Verification of genetic purity of remnant cutthroat trout populations is essential in restoration efforts. Genetic purity of native cutthroat trout cannot be visually ascertained with certainty because hybridization can be minor and not phenotypically expressed. Also, Bonneville cutthroat trout evolved from multiple origins (polyphyletic) and are represented by several genetically diverse groups over a relatively wide geographic area (Hickman and Duff 1978, Martin et al. 1985, Behnke 1992, Shiozawa and Evans 1994). Northern forms of Bonneville cutthroat trout from Bear Lake in Utah-Idaho and the Bear River in Utah-Idaho-Wyoming likely evolved from a relatively recent ancestral salmonid that invaded the ancient lake; they remained partially isolated from more southern portions of the basin because Bear Lake and the Bear River were large systems lying outside of the once-inundated prehistoric Lake Bonneville. Similarly, both Bonneville cutthroat trout from the Deep Creek mountains in Utah's west desert and those from the extreme southern portion of the basin in southern Utah remain somewhat genetically distinct, despite being taxonomically classified as a single subspecies.

At present UDWR minimum standards for genetic testing of each new population includes meristic analysis from a random sample of 10 fish, DNA analysis (mitochondrial and nuclear) from 30 fish, and consideration of geographic location and historical stocking records. Also, a method to quantify introgression has been established with "core populations" defined as those with <1 percent introgression, suitable for restoration of new populations, and "conservation populations" defined as those with <10 percent introgression, designated for continued preservation (Utah Division of Wildlife Resources 2000).

Despite taxonomic complications, even the earliest putative identifications based on visual appearance (field observations), capture location, and stocking history were generally accurate, based on subsequent independent expert verification based on meristic, allozyme, or molecular DNA analyses (Table 1). Most remnant populations of native trout in southern Utah were each evaluated with different genetic tests over time. Some populations were tested as many as four times over a 20-year period. Although tests became increasingly more sophisticated, all populations originally given "conservation" status as far back as the 1970s have remained as such.

In two instances results changed over time with repeated genetic testing. The headwater population of cutthroat trout in the North Fork North Creek (Table 1) was originally suspected to be hybridized because of its proximity to rainbow trout in downstream reaches and the absense of a barrier separating the two species. Early genetic tests (meristics 1981 and allozymes 1981) did not show any hybridization, and actions were taken to prevent upstream movement of rainbow trout; however, 2001 test results (nuclear DNA) showed some rainbow trout hybridization (6.4%). Regardless, levels were low enough and within established standards to still allow a designated "conservation" status. Another population (Deep Creek) originally thought to be hybridized, due to contamination of test samples, was later determined to be pure. This mistake caused delays in replicating the Deep Creek population in other locations and resulted in continued skepticism about genetic purity because of early reports that the fish were introgressed (Behnke 1976).

Researchers in locations outside of southern Utah reported cases where only slightly introgressed populations were found after initial evaluations suggested a high probability of introgression. Neilson and Lentsch (1988) reported that hybridization of Bonneville cutthroat trout from Bear Lake, Utah-Idaho, was only minute despite long-term stocking of rainbow trout (*O. mykiss*). Gamblin et al. (2000) originally found high percentages of rainbow trout hybridization with Yellowstone cutthroat trout (*O. c. bouvieri*) in Henrys Lake, Idaho; however, later testing documented spawning runs of cutthroat trout with little introgression.

Management decisions were based on the best genetic techniques available at any given time. However, time to complete tests, gain clearances for field projects, and actually conduct projects was so great that new methodologies for testing would often evolve before projects were complete. Thus, there was often concern that populations should be re-tested with new techniques to make sure earlier tests were accurate. Potentially, this can cause delays in completing restoration projects and sway emphasis of work towards more genetic testing. For example, development of a wild brood stock of Colorado River cutthroat trout from Boulder Creek was initiated based on mitochondrial DNA tests conducted between 1990 and 1993 (Table 1). It took several years to complete genetic tests, obtain disease clearances, transplant trout to a suitable lake, and eventually take eggs from spawning trout. By 1999 when the first eggs were ready to be cultured, new nuclear DNA tests had been developed and agency personnel questioned if the project should proceed without confirmation of genetic status with newer more sophisticated tests. Even in this case, re-testing would have been feasible if conducted in a timely manner, but long turnovers in laboratory times, a state-wide backlog in test samples, and acquiring funding to complete tests has resulted in periods of 1-2 years or more to complete tests. Rather than delay work, the Boulder Creek project proceeded based on available information. At times, balance needs to be achieved in accepting some risk by using older test results, by conducting field projects in a timely manner, and by deciding if re-testing populations with new techniques is warranted.

Success of Using Rotenone

As part of native trout restoration, we

renovated 29 streams with rotenone during the 25-year period under review (Table 2). Renovated streams were relatively small; the largest (UM Creek) did not exceed a base flow of 0.4 m^3/sec and included 37 km of main stem and tributaries. Even when treating the smallest streams, a one-time treatment with rotenone apparently could fail to remove all nonnative fishes. Trout spawning sites associated with springs and seeps presented the greatest difficulties; these areas provided freshwater refuges for small fish where rotenone failed to make contact. Often when trout were missed with a single application, young-of-the-year or eggs persisted. Second treatments, timed approximately a year after the first treatment, generally completely eradicated target species.

Sam Stowe Creek was the first renovation project conducted in 1977, consisting of a simple 4.8-km first-order stream, successfully completed with a one-time treatment. The second project, conducted on another first-order 5.0-km stream (Pine Creek) in 1980, failed to totally remove rainbow trout, which led to second treatments as a standard practice. For all projects first-year application of rotenone was approximately 50 percent successful in completely eliminating target species; nonnative trout were found about half the time with second treatments. Rainbow trout, as well as brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) were target species that were often missed.

UM Creek represented a unique situation where treatments were conducted on four consecutive years in attempt to completely remove brook trout. All other projects were completed in 2 years. The project was designed to protect downstream state and private fish hatcheries from whirling disease but also presented an opportunity for restoration of native trout. Similar to many other projects, large numbers of young-of-the-year brook trout were found during the second treatment where adults had been observed spawning the previous year. Although brook trout were removed from most of the drainage after the second treatment, they persisted in one spring until the fourth treatment. Brook trout avoided rotenone in this spring by moving less than a meter into underground caverns. Treatments were not effective until a combination of rotenone and electrofishing gear was used at this site. Such an effort might have been effective after the second treatment, but it took 4 years to become familiar with the entire drainage and identify problem areas. Extensive post-treatment surveys of UM Creek conducted over 6 years have failed to find any additional brook trout. In fact, all treatment projects conducted with rotenone in southern Utah were successful in completely removing nonnative trout after multiple treatments.

Sources of Native Trout

In the late 1970s and throughout the 1980s little concern was expressed about origin and destination sites for relocating native trout. We avoided later criticism regarding indiscriminate movements by fortuitously selecting sources of native fish for re-introductions from sites in close proximity, even before more sophisticated information regarding localized and regional genetic differences in fish populations was available. For example, several streams were selected for Bonneville cutthroat trout restoration projects during the 1980s in the Virgin River drainage (Fig. 1 and Table 2). Bonneville cutthroat trout from the Sevier River drainage could easily have been used as source fish, resulting in an inter-basin transplant; however, local fish from other Virgin River tributaries were used. Local fish were a more practical choice because they were suspected to be native to this area, despite being found just outside the Bonneville basin (Hepworth et al. 1997a). By the 1990s, after inter-agency conservation team planning was established, concepts of utilizing the closest available source of native trout within the same drainage (nearest neighbor) became a standardized practice.

Defining "conservation" and "sport fish" populations in multi-agency conservation agreements helped set formal limitations on fish transplants (Lentsch and Converse 1997, Lentsch et al. 1997). Although both conservation and sport fish designations for cutthroat trout populations usually allow legalized sport angling, conservation populations (<10% introgressed) are those managed specifically with naturally reproducing wild trout to maintain genetic integrity of the subspecies. As previously noted, the definition of a conservation population was eventually subdivided to include "core" populations (<1% introgressed) designated as suitable to replicate in other areas, generally on a "nearest neighbor" basis (Utah Division of Wlidlife Resources 2000). Sport fish populations are defined to include areas managed by stocking native trout produced in state hatcheries to maintain public sport fisheries where limited or no natural reproduction occurs. Providing that stocking is not a threat to conservation populations, it can take place over a wider geographic area compared to the more restricted "nearest neighbor" concept. Sport fish populations might consist of 100 percent hatchery fish that could be genetically pure native cutthroat trout. The definition of sport fish populations of native cutthroat trout also includes wild populations that are >10 percent introgressed with nonnative trout.

Designating "geographic management units" in conservation strategies (Lentsch and Converse 1997, Lentsch et al. 1997) also encouraged transplants, stockings, and wild brood stock development to occur within the bounds of natural watersheds, avoiding inter-basin fish transfers. The Southern Geographic Management Unit for Bonneville cutthroat trout included the Sevier River drainage and a small portion of the Virgin River drainage (Fig.1), but even within these areas the proximity of sub-drainages and individual streams were considered when making transplants. For Colorado River cutthroat trout, the Southern Geographic Management Unit consisted of the Escalante and Fremont river drainage. Since no remnant populations of native trout have been found or are likely to be found in the Fremont River drainage, native trout from the Escalante River drainage were used to restore populations in the Fremont River drainage because they were the "nearest neighbor."

Wild brood stocks of both Bonneville and Colorado River cutthroat trout were established at Manning Meadow Reservoir and Dougherty Lake, respectively, for sport fish and conservation management purposes (Table 2). Brood stocks were created from multiple stream sources within respective geographic management units to maximize the initial size of the transplanted populations, increase genetic diversity, and avoid bias from over-use of any single fragmented population that might not be representative of native fish from the overall geographic management area. Most restoration projects were conducted without using fish produced from wild brood stocks by transplanting individuals from core populations to establish conservation, or core, populations in other locations (Table 2). Nevertheless, some restoration projects were not feasible without a brood stock of native trout. For example, UM Creek involved a large area and required a relatively short period of time between removal of nonnative trout and re-establishment of sport fishing opportunities in order to avoid a significant public controversy. Transplanting limited numbers of wild trout could not satisfy recreational demands. Thus, hatchery trout produced from wild brood stock were used. In other cases lakes and reservoirs that required stocking for sport fishing purposes were connected to wild trout streams (Barney and Manning reservoirs – Manning Creek; Willow Bottom lakes – Twitchell Creek; Robs Reservoir – Center Creek; Pine Creek Reservoir – Pine Creek; Forsyth Reservoir – UM Creek; Table 2). Such areas can be restored if stocking relatively large numbers of native trout is an option. Once all nonnative trout are removed, lakes dependent on stocking can be maintained by stocking native trout while streams become self-sustaining with native trout. This also creates the potential to enhance spawning habitat for lake populations, if possible, eventually managing lakes entirely with wild trout.

In a few cases, sterile tiger trout (*Salmo trutta X Salvelinus fontinalis*) were stocked after rotenone treatments to replace popular sport fisheries and then phased-out as re-introduced native trout expanded from natural reproduction. Tiger trout were used in Manning Meadow and Barney reservoirs, and UM Creek (Table 2). Growth, survival, and catchability of tiger trout was sufficient to produce sport fisheries in both reservoirs and streams. When tiger trout were first produced in Utah in the 1990s, it appeared availability would be limited because of difficulty in culturing large numbers of fish. Egg survival was typically about 4 percent; however, subjecting eggs from this hybrid cross to a hot water bath producing triploids increased egg survival to more normal production rates of 70-80 percent (Scheerer and Thorgaard 1983) and allowed greatly expanded use of these fish. Stocking rates were monitored and adjusted to avoid excessive competition and predation between tiger trout and native trout and then discontinued as native trout became available for stocking or as native trout naturally expanded into areas where tiger trout had been stocked. The greatest problem encountered with tiger trout was that they often became more popular with anglers than native cutthroat trout. Public pressure was exerted in several situations to maintain stocking of tiger trout, in contrast to conservation plans that emphasized native trout. We partially alleviated this problem by shifting tiger trout stocking to other sites.

We limit the number of native trout transplanted from source populations to protect these areas. The number of fish transplanted is based on size of the source population and its ability to replenish itself. We limit the number of fish transplanted, leaving behind young-of-the-year and large adult fish, by taking a wide variety of intermediate ages and sizes (both adults and sub-adults). Also, a portion of each source population is set aside as a refuge area from which transplanted fish are not collected. Less than 20 percent of the stream length of a source population is subjected to removal of fish in any single year. These conservative guidelines are partly based on an experience where 1024 Bonneville cutthroat trout were removed from Pine Creek (Table 2) between 1988 and 1991; the population was affected by removals but ultimately recovered. Pine Creek, a 5-km stream, was originally restored in 1980 with the intent to increase numbers of Bonneville cutthroat trout to provide fish for a wild brood stock. By 1984, Pine Creek contained 298 cutthroat trout/km. State policy governing state fish hatcheries and wild brood stocks require 3 years of disease certification before moving fish into or out of these locations. Certification was completed on Pine Creek by sacrificing 120 Bonneville cutthroat trout annually from 1988 to 1990 for disease tests. In addition, 469 and 245 fish were transplanted from Pine Creek to Manning Meadow Reservoir in 1990 and 1991, respectively. By 1991 it was apparent that the Pine Creek population was suppressed, as the stream distance and effort needed to collect 245 fish was far greater than what was needed for equivalent collections in previous years. Nevertheless, by 1995 (Hepworth et al. 1997b) the population had recovered to 228 cutthroat trout/ km (270 in 2001) with numerous sizes and ages of fish. Despite the high number of fish removed and temporary reduced population size, there did not appear to be a long-term affect.

Restored streams generally received a minimum of 100 fish transplanted from core populations and >200 if possible. These values originally resulted from number of fish available from core populations and time required for transplanted fish to re-populate renovated streams. Sam Stowe Creek was restocked with 39 fish in 1977 (Table 2), and after 7 years the full 4.8 km of available stream habitat was not fully repopulated. In comparison, Pine Creek (5 km) was restocked with 245 fish in 1981, which repopulated all available habitat within 3 years. Transplanted fish were a mixture of adults (≥age-3) and sub-adults, but most fish were likely mature within a year after being moved. In some cases we transplanted fish over several years to increase overall numbers. Although we did not specifically base number of transplanted fish on genetics and effective population size, i.e., number of breeding adults within a population, numbers were likely sufficient to prevent genetic drift and inbreeding depression. Franklin (1980) suggested that an effective short-term population size of 50 was sufficient to prevent loss of genetic diversity in small populations if in the long-term effective population size expanded to at least 500.

Migration Barriers

Naturally existing fish migration barriers protected most of the remnant populations of native trout in headwater streams in southern Utah and were responsible for their persistence (Table 1). Of 11 remnant populations, seven were isolated by multiple natural waterfalls or a combination of waterfalls and naturally intermittent stream sections. A hydroelectric power diversion that created a barrier falls and a de-watered section of stream isolated another remnant population in West Boulder Creek. Two remnant populations (East Boulder Creek and White Creek) persist in remote locations upstream from simple single-point barrier falls with populations of nonnative trout directly downstream. The North Fork North Creek and West Pine Creek were the only two remnant conservation populations that persisted in headwater areas without obvious physical barriers and in contact with nonnative trout. Some form of physiological attribute likely allowed native trout to retain a competitive advantage over nonnative trout in these areas, but in the case of the North Fork North Creek, minor introgression occurred. The West Pine Creek population persisted in the presence of brown trout and was not threatened with hybridization (Hepworth et al. 2001).

As such, artificial barriers were deemed necessary to protect restored populations of native trout (Table 2). Barriers can, however, potentially create problems by increasing fragmentation and limiting natural fish migrations (Kershner 1995, Young 1995b). Rather than further fragment existing native trout populations, we used barriers as part of restoration projects to increase the range of native trout and decrease fragmentation. Populations of native trout were restored in areas where they had been totally extirpated; barriers used to isolate these locations from nonnative trout. In other situations, remnant populations were expanded by constructing barriers at downstream locations, removing nonnative trout above the barriers, and allowing headwater populations of remnant native trout to expand into the renovated stream sections. Plans include reconnecting the remnant population in West Pine Creek with a putative population in North Pine Creek, thus reducing fragmentation by use of barriers constructed near the lower end of the main stream. Similarly, other restoration projects included second and third order streams with multiple tributaries (Table 2) whereas remnant populations of native trout had been restricted to simple first order streams (Table 1).

Most barriers were constructed of large, selectively placed rock to form check dams and waterfalls of at least 1.5 m. Road culverts were used to do the same thing when projects could be coordinated with road work in suitable areas. Over time, it became apparent that effective barriers required splash pads to prevent formation of plunge pools at the base of the falls, thus limiting the ability of trout to jump barriers.

Barriers that worked best were adjacent to other obstacles that limited fish movement such as seasonally de-watered stream segments (Birch Creek, Table 1; Sam Stow Creek, Manning Meadow Creek, Tenmile Creek, and Pine Creek, Table 2). The only barrier constructed as a single-point structure, where nonnative fishes had a continual presence immediately downstream, partially failed because of formation of a plunge pool (North Fork North Creek; Table 2). The remnant headwater population of native trout in the North Fork North Creek was likely conserved although efforts to expand these fish were only partially successful. Although fish above the barrier phenotypically appeared to be cutthroat trout, genetic tests conducted in 2001 found fish from the restored stream section to be more introgressed (15%) than headwater fish (6%).

We relied on an intermittent stream section and a road culvert to function as barriers for the original 1977 renovation of Sam Stowe Creek. Road construction in the 1980s altered the culvert, and high spring flows in the early 1990s allowed rainbow trout to migrate into this stream even though it was thought to be isolated. Another barrier had to be constructed and the entire project was re-conducted in 1997.

Brown trout regained access into lower Threemile Creek as a result of high water flows through a normally de-watered stream section. These fish did not, however, move past a barrier that was constructed as part of the 1994 renovation project, and brown trout were selectively removed from the lower stream segment by electrofishing 500 m of stream.

In cases where construction of barriers with secondary obstacles was not possible, we opted in recent years to construct multiple barrier waterfalls (West Boulder Creek, Pine Creek, and White Creek, Table 2), similar to many natural situations where multiple obstacles protected remnant populations. Multiple barriers created a buffer zone that could be easily monitored and readily renovated should nonnative fish gain access above the lowest barrier and assured overall project success should a single barrier fail.

In general, barriers should be considered temporary or constructed with an understanding that they will likely require long-term maintenance. Given enough time without maintenance, they will likely fail. Even if barriers are needed for 15-20 years, long-term plans should focus on their elimination by expanding populations within larger portions of overall watersheds, providing that both biological and socio-political solutions can be satisfied.

Restoration of Metapopulations

Conceptually, metapopulations have a greater probability of long-term persistence than smaller populations. Cutthroat trout metapopulations have greater demographic stability than smaller populations; they allow large-scale fish movement and migrations, interconnect smaller populations, allow replacement after stochastic or catastrophic losses of fish from individual streams, and provide for large and diverse gene pools (Kershner 1995, Young 1995a, 1995b). Thus, restoration efforts logically would attempt to restore native trout over larger areas; however, we found that larger and more complex restoration projects that had metapopulation characteristics were subject to more potential problems and failures than smaller projects. This discrepancy resulted from complications of human impacts on natural systems and because of greater difficulty removing all nonnative fishes with rotenone in larger systems.

Metapopulation function and theory is often viewed as it applies to natural situations without regard to human influence, but human impacts cannot be overlooked in practical management situations. Gresswell et al. (1994) found numerous human impacts affected the Yellowstone Lake metapopulation of cutthroat trout even with park protection and noted that a single illegal introduction of rainbow trout could threaten this entire complex of native trout. Mangel and Tier (1994) explained that risk of extirpation from catastrophic events can be as high for large populations as it is for small populations, and that corridors connecting populations can provide pathways for catastrophic losses and extinction. This was especially true for cutthroat trout, given that the foremost factor credited to population declines in the late 1800s and early 1900s was the introduction of nonnative trouts (Behnke 1992, Kershner 1995, Young 1995b, Hepworth et al. 2001). These nonnative fishes spread throughout interconnected waterways and replaced native cutthroat trout, including native trout in the major river systems in southern Utah (Popov and Low 1950, Cope 1955).

By 1977 the only native trout remaining were remnant populations restricted to first order streams composed of headwater tributaries to the larger rivers. We restored native cutthroat trout populations in relatively simple stream systems in the 1970s and early 1980s (Table 2) and then progressed into larger drainages with multiple tributaries (South Ash Creek, Leeds Creek, Threemile Creek) starting in the late 1980s as opportunities and methodologies allowed. By the 1990s with development of conservation strategies, restoration goals expanded to include re-establishing native trout in larger and more complex systems, if possible, with increased connectivity and at least some characteristics of metapopulations. By the late 1990s we began restoring multiple tributary streams interconnected with lakes and reservoirs (Manning Creek, Manning Reservoir and Barney Reservoir; UM Creek and Forsyth Reservoir). These larger systems were restored systematically as a combination of smaller projects completed over multiple years. At the same time we continued to restore smaller, first order streams (Birch Creek-B, Tenmile Creek, and White Creek).

In our evaluation small projects conducted in fragmented streams were less subject to negative interventions by man compared to larger systems. Illegal movement of nonnative fish was not a problem while working with small isolated streams that were of little interest to the public for sport fishing. Conversely, this risk and difficulty of removing nonnative fishes increased as restoration projects expanded into larger areas with greater amounts of sport fishing. Although most restoration projects remain successful, public complaints have been common and requests have been made to stock nonnative trout in UM Creek, Dougherty Lake, and Manning Meadow Reservoir. Most southern Utah reservoirs >100 ha have had illegal fish introductions within the last 25 years.

Managers also should consider threats to native trout in regard to time and set management priorities on that basis. It does little good to attempt to prevent long-term threats such as inbreeding depression by establishing metapopulations if short-term problems such as expansions of nonnative fishes are not dealt with first. This can be true even if the short-term actions appear temporarily detrimental to long-term considerations.

We suggest restoring native trout in multiple historic sites, working with both large and small-scale systems. Managers should consider a variety of projects realizing that threats can be both natural and human-caused, and that project feasibility and potential for success can vary in different situations. For example, *conservation* of an existing metapopulation might be a high management priority regardless of the risks. In contrast, complete *restoration* of a metapopulation might not be justified although some attributes of a large interconnected system appear attractive. Even if high risk is only associated with a single threat, time and agency resources put in jeopardy by such a risk might be too great to justify the project under these conditions.

Socio-political Issues

Obtaining regulatory clearances to conduct recovery projects has increasingly become more complex and difficult. In the 1970s restoration projects received little resistance or concern, but there was little funding and few programs to support such work. Increased awareness that a subspecies of cutthroat trout could be listed under the ESA and the establishment of state and USDA Forest Service sensitive species lists helped establish funding mechanisms and justify expenditures on programmatic approaches to conservation. At the same time, these actions had some negative affects. Opposition developed to transplanting cutthroat trout because of their sensitive status and the associated implications they might have on other land management issues. Restoration projects on Spring Creek in 1993 and Deer Creek in 1994 were canceled after considerable planning effort because of conflicting issues with other land uses and promises UDWR had made with the local counties to avoid such problems. Even while the USDA Forest Service officially supported native trout restoration through conservation agreements, some district rangers and land use specialists opposed transplants simply to avoid associated complications and controversy. More importantly, county governments expressed opposition to expansions of native trout because of possibile federal listing and subsequent restrictions on resource use. At times local governments were often skeptical of state agency objectives, wondering whether we wanted to prevent federal listing or rather wanted sensitive species to drive land use policy. Over time, obtaining regulatory clearance for use of rotenone has become more difficult. It remains unclear how the National Environmental Policy Act (NEPA) applies to state restoration projects conducted on federal lands. Inconsistencies in implementing NEPA processes persist among agencies and even among National Forests.

To help alleviate concerns over sensitive species listings, inter-agency conservation agreements for native cutthroat trout in Utah were established in 1997 among the UDWR, Utah Department of Natural Resources, Utah Reclamation Mitigation and Conservation Commission, USDI Fish and Wildlife Service, USDA Forest Service, USDI Bureau of Land Management, and USDI Bureau of Reclamation (Lentsch and Converse 1997, Lentsch et al. 1997). The objective of the agreements was to remove threats to cutthroat trout that could lead to federal listing under the ESA. The agreements reduced local concerns over the sensitive status of cutthroat trout by allowing transplants and introductions to take place without giving new populations sensitive species status; naturally existing populations and populations established prior to the agreements retained sensitive status. However, about the time that conservation agreements were being finalized and implemented, local government concerns culminated with passage of a 1998 state law in Utah requiring county approval of written plans for any transplants of state or federal sensitive species (1998 Utah Code 23-14-21). In addition, the Endangered Species Protection Fund was established in Utah the same year, sponsored by rural legislatures (1998 Utah Code 63-34-14). Its purpose, in part, is to provide funding to the State to pro-actively manage sensitive species, thus preventing the need for federal listing under ESA.

To comply with the transplant law, recent restoration projects (Table 2) were planned by the UDWR and then approved by local counties. The development of county approved plans was incorporated into the NEPA process and includes writing an Environmental Assessment although an actual need for this level of NEPA compliance remains unclear. Cost of project planning and approval has greatly increased and is generally much greater than the cost of actually completing field work. It is no longer cost and time effective to work on plans for small individual streams because small projects require as much planning effort as large projects. We have partially solved this problem by developing Environmental Assessments that include restoration plans for up to 10 or more lakes and streams.

To date, the new transplant law has not resulted in project terminations. Counties have shown a willingness to grant approvals, knowing they now have considerable authority in the process and knowing that projects will be terminated if listing status changes, which would void current plans. For instance, stocking of Bear Lake Bonneville cutthroat trout by the state of Utah would have to be discontinued in Bear Lake, Utah-Idaho according to the 1998 Utah law if Bonneville cutthroat trout were listed under the ESA – at least until a new plan was developed and approved by the local county. As a result, federal listing poses a more serious predicament for state managers than it did prior to the 1998 Utah law. Behnke (1992) cautioned that compromise needs to be reached in application of the ESA to avoid a public backlash against the act and not have conservation efforts immobilized. Federal listing of Bonneville and Colorado River cutthroat trout under the ESA could stop, at least in Utah, culture from wild brood stocks, annual stockings, annual introductions of millions of native trout, and millions of dollars of management and hatchery programs that might be difficult to reinstate.

Ironically, the greatest remaining threat to continued native trout conservation in Utah by UDWR could be the ESA. Although federal law would pre-empt state law, it might not be an issue. The 1998 state law simply requires local government participation in the planning process for restoration projects. Under current conditions, cooperation has evolved among local, state, and federal agencies because of a mutual goal to pro-actively prevent federal listing. Considerable cooperation and incentive among agencies could be lost as a result of federal listing and replaced with mandates attempting to force conservation, which would likely meet local resistance and an even larger adversarial presence in development of conservation plans. Granted, in other situations where management actions for a species might be lagging, listing under ESA could be the most appropriate method to stimulate agency response.

More recently, as restoration projects were expanded into larger and more complex systems, anglers expressed increased concern over potential loss of nonnative but popular sport fisheries. In response, we have stressed the importance of using native trout in ways to improve angling (Hepworth et al. 1999, 2000). In one case, native cutthroat trout stocked in a small reservoir were found to have higher over-winter survival than rainbow trout. In other situations, stunted brook trout in small lakes are being replaced with native cutthroat trout that will attain larger sizes and be more attractive to anglers (Willow Bottom lakes, Robs Reservoir, and Pine Creek Reservoir, Table 2). Projects should be planned to increase support for native trout, rather than create public opposition.

Conclusions

We found by trial and error that restoration projects fail because of various reasons. As previously discussed, problems were encountered with incomplete removal of nonnative trout with rotenone, barrier failures, and potentially from surreptitious stockings. Another factor that can influence project success is habitat suitability. Durfey Creek (Table 2) was fish-less prior to introducing Colorado River cutthroat trout. The introduction likely failed because of cold water temperatures which did not exceed 10°C. Some of the transplanted fish persisted for at least 3 years but failed to reproduce. Harig et al. (2000) described failure rates >50 percent for transplanted populations of native trout in Colorado and New Mexico streams due to low water temperature, small stream size, and degraded habitat. We evaluated habitat suitability for restoration sites based on present conditions for nonnative trout and the histories of trout in these streams. Most restoration projects we conducted were selected because they had habitat that supported healthy nonnative trout populations. The USDA Forest Service recommended Threemile Creek as a restoration site because changes in livestock grazing had already been made without native fish being an issue. Riparian habitat and stream conditions had improved prior to restoring native trout in 1994.

In other situations we discontinued restoration projects on Spring Creek in 1993 and Deer Creek in 1994 (Table 2) because of socio-political issues. Livestock grazing apparently was going to become an issue on

Spring Creek, and we had committed to local government officials that restoration work would be conducted in a non-controversial manner. At Deer Creek it became apparent that it would be difficult to remove all nonnative trout with rotenone, stay within the defined project area, and not cause a controversy with local sport fisheries. Although failure of these latter two projects was disappointing, it afterwards added credibility to overall restoration efforts when additional project approvals were sought and we claimed that conflicts with other land uses would be avoided as they had been in the past.

Overall, we found five important factors in selecting sites for restoring populations of native trout. Projects should:

(1) have habitat capable of supporting multiple year-classes of wild trout over many years;

(2) be cost and time effective in regard to the size of the restoration project and justify renovation efforts;

(3) be feasible by having a high probability that all nonnative fishes can be removed with rotenone and prevented from returning;

(4) avoid major land use conflicts; and

(5) have support from the public and land management agencies.

Although finding a perfect restoration site is difficult in light of the above factors, advantages can be weighed against disadvantages. For any given location, potential problems can be recognized and pro-active plans made to deal with these concerns or avoid problems by selecting an alternate site.

Success and failure rates are not a valid means to evaluate overall restoration success. Potential restoration projects often arise from circumstances other than those considered primarily for native trout. UM Creek was restored due to efforts to control whirling disease (Table 2). Leeds Creek and Birch Creek were restored because wild fires resulted in large losses of nonnative trout present at the time of the fires. Durfey Creek was selected as a transplant site because it was a fish-less stream where a transplant could be conducted without approval to conduct a rotenone treatment. Although habitat at some sites might be marginal or other problems such as whirling disease might be present, an opportunistic restoration attempt does little harm as long as fish are available for transplant without jeopardizing source populations. Greater risk in some situations might be associated with the potential for greater gain. It is important to understand why projects fail, but keeping score of failure rates has little overall value. Total restoration progress gained versus time and money spent is a more useful way to evaluate success.

Maintaining successfully restored streams and lakes requires frequent monitoring and a long-term commitment. We scheduled population surveys on all native trout streams every 7 years. This allowed time to conduct restoration projects between monitoring populations, which in turn, gave direction in planning new restoration projects. Nevertheless, we suggest more frequent spot-checks to survey key areas, check recently treated areas, maintain migration barriers, monitor habitat conditions, and evaluate re-colonization of restored areas by recently introduced fish. Some of these concerns should be evaluated annually in recently restored areas, then evaluated less frequently after it is determined that projects were initially successful and as such, reached a higher level of security. For example, UM Creek has been monitored annually since 1996 to evaluate progress as the native trout population develops, while South Ash Creek was restored in 1986, watched for a few years, and then surveyed in 1995 and 2002. Unless some problem becomes evident, South Ash Creek will likely not be surveyed again until 2009.

Altogether, restoration projects were conducted on 42 streams and lakes within southern Utah during the past 25 years (Table 2). Seventeen of these projects have been successful in establishing self-sustaining populations from the time the projects were first completed. Ten projects were conditionally or partially successful after supplemental actions were taken to overcome problems that occurred subsequent to the initial restoration effort. Three restoration projects failed and there was no other attempt to restore these sites. An additional 12 restoration projects are still in progress where self-sustaining populations are expected to become established. Despite some problems and delays, native trout increased from just three remnant populations in about 9.9 km of stream known to occur in 1977, to established populations in 33 streams and over 150 km by the year 2002. Restoration projects in progress, if successful, will increase stream habitat by another 105 km and lake habitat to a total of 108 ha (9 lakes and reservoirs) managed as native trout conservation populations. Additional projects are planned for the future. In addition, expanded use of native cutthroat trout produced from wild brood stocks was developed for general sport fish management applications. Stocking of all nonnative cutthroat trout was discontinued in Utah after 1999. Native trout restoration has been successful because it was an important part of the state's annual work plan with a dependable budget, interest and commitment increased among other resource agencies, and the ESA posed a common objective among agencies and local governments to prevent listing.

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Further Evidence That Lake Trout Displace Bull Trout In Mountain Lakes

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Abstract

I surveyed five large mountain lakes in Glacier National Park, Montana, with gill nets in 2000 to assess the status of bull trout (*Salvelinus confluentus*) populations. I compared results to previous surveys, conducted in 1969 and 1977, at which time numbers of native bull trout were higher than recently established populations of lake trout (*Salvelinus namaycush*). The data indicate a broad decline in bull trout numbers and corresponding increases in lake trout population size in Kintla, Bowman, Logging, and McDonald lakes. In Quartz Lake, where lake trout are not known to occur, bull trout catch was stable across years. These data suggest that lake trout expansion has had a substantial detrimental impact on Glacier National Park bull trout populations especially because variables commonly implicated in bull trout population decline elsewhere across the species' range are not significant factors in Park lakes. I contend that effective recovery actions for adfluvial bull trout populations, in mountain lakes where nonnative lake trout have become established, must be directed at reducing species interaction through directed control actions on lake trout. I suggest that the rate and magnitude of the transition from native bull trout to introduced lake trout may depend on multiple factors, including migration of either species, the extent and quality of bull trout spawning and rearing habitat, and the structure of the lacustrine food chain. Four of the five bull trout populations I studied in Glacier National Park lakes are currently at high risk of extirpation, due primarily to incompatibility with introduced lake trout populations.

Key words: bull trout, introduced, lake trout, mountain lakes, nonnative, population

Introduction

In 1998 bull trout (*Salvelinus confluentus*) in the Columbia River basin were listed as a threatened species under the U.S. Endangered Species Act (USDI Fish and Wildlife Service 1998). Since that time there has been an increased emphasis on determining their distribution, abundance, and genetic status. Status information is a critical component of the federal bull trout recovery planning process (Lohr et al. 2001).

The Montana Bull Trout Scientific Group (MBTSG) identified evaluation of the status of bull trout in lakes on the west side of Glacier National Park (Park) as a priority research need (MBTSG 1995). This study lies within the Flathead River watershed, part of the headwaters of the Columbia River basin in northwestern Montana (Fig. 1). Bull trout occur in 16 lakes across 10 drainages within the Flathead River watershed in the Park (MBTSG 1995). The Montana Bull Trout Scientific Group (MBTSG 1995) considers these populations to be disjunct, suggesting they are located in headwaters lakes that are reproductively isolated from the downstream population in Flathead Lake.

Lakes in the Flathead River drainage within the Park support a low diversity of native fish species, probably because of incomplete postglacial recolonization from downstream. Flathead Lake, itself the largest natural freshwater lake in the western United States and a source for postglacial dispersal, contained only 10 native fish species (Spencer et al. 1991). A typical native species assemblage in Park lakes west of the Continental Divide consists of bull trout, westslope cutthroat trout (*Oncorhynchus clarki lewisi*), mountain whitefish (*Prosopium williamsoni*), longnose sucker (*Catostomus*

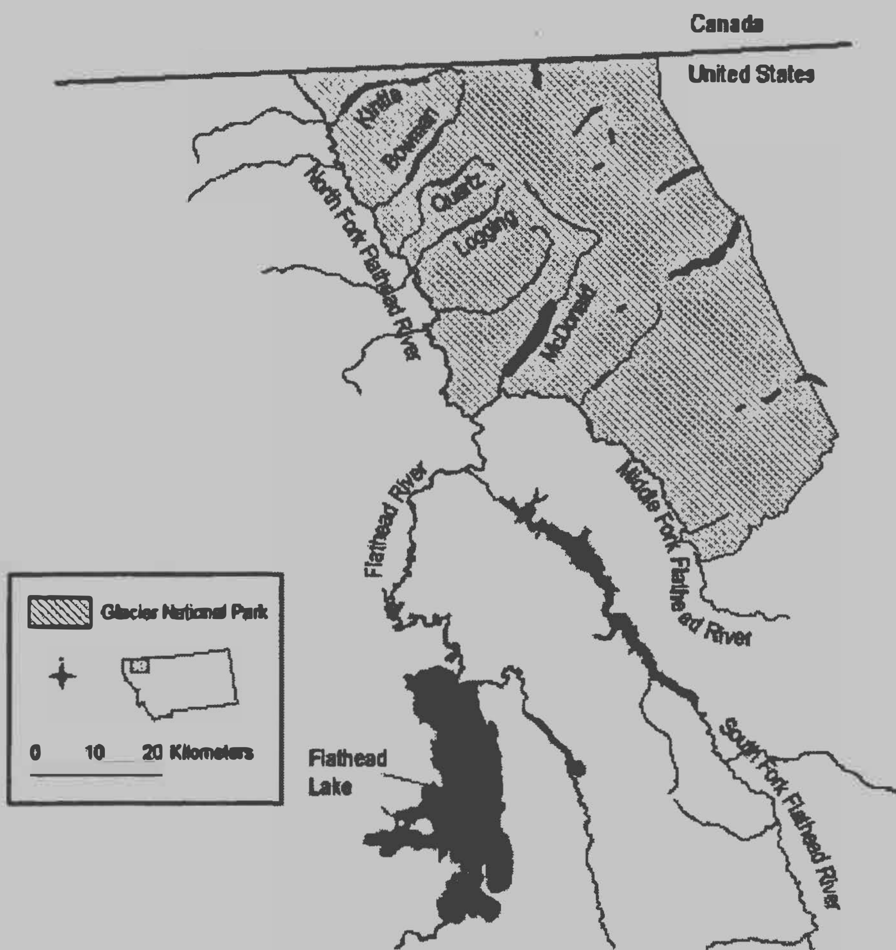

Figure 1. Map of study area, showing lakes in Glacier National Park which were surveyed in 2000.

catostomus), largescale sucker (*Catostomus macrocheilus*), and slimy sculpin (*Cottus cognatus*). Native cyprinid species including northern pikeminnow (*Ptychocheilus oregonensis*), peamouth (*Mylocheilus caurinus*), and redside shiner (*Richardsonius balteatus*) as well as pygmy whitefish (*Prosopium coulteri*) exhibit spotty distribution in these large glaciated lakes. In several of the lakes nonnative kokanee (*Oncorhynchus nerka*), brook trout (*Salvelinus fontinalis*), or rainbow trout (*Oncorhynchus mykiss*) occur, though none of these are abundant. In this ecosystem bull trout is the only native fish species that is highly piscivorous.

Lake trout are native only to the Saint Mary River drainage on the east side of the Park in the headwaters to the Hudson Bay drainage. There is no historical documentation of the intentional introduction of lake trout into any Park lakes in the Flathead River drainage (Morton 1968a, 1968b). The initial introduction of lake trout outside the Park in the Flathead drainage is believed to have been carried out by the U.S. Fish Commission in 1905 (Spencer et al. 1991), leading to establishment of this species in Flathead Lake.

Telemetry studies have illustrated the mobility of lake trout, e.g., fish tagged in the Flathead River system ranged through most accessible waters, including possible movement upstream into Lake McDonald in the Park (Muhlfeld et al. 2000). The authors surmised that lake trout movements could be related to water temperature, stream flow, and food availability. Seasonally cold water temperatures, e.g. in early summer, in streams emanating from headwater lakes may provide attractive thermal refuge for migrating lake trout in the Flathead River, offering one possible explanation for lake trout invasion of Park lakes.

Unrecorded stocking or illegal transplants cannot be ruled out as original sources of lake trout that colonized Park waters, but there is no anecdotal or documented verification of either case. Obtaining and transporting small lake trout for live transplant, particularly several decades ago, would have been exceedingly difficult. Natural migration of fish from Flathead Lake is a more likely source of these populations.

Regardless of the mechanism of introduction, over the past 50 years lake trout have become established in most of the larger Park lakes in the North Fork Flathead and Middle Fork Flathead River drainages. With their naturalization has come potential impacts to native species. Donald and Alger (1993) studied the interaction between lake trout and bull trout in mountain lakes of the Rocky Mountain region of southern Alberta and British Columbia into northwest Montana. They documented substantial niche overlap in which lake trout dominate. They concluded that lacustrine populations of bull trout usually cannot be maintained if lake trout are introduced. Donald and Stelfox (1997) recommended that stocking of other *Salvelinus* species not occur in waters where the objective was to maintain adfluvial populations of bull trout.

Where established lake trout populations exist in Park waters west of the divide, an abundance of anecdotal evidence (Glacier National Park, West Glacier, unpublished data and file reports) suggests that the number of bull trout present has declined over the past 25 years. Because of recent concern for bull trout, the primary objective of my study was to document temporal changes in bull trout abundance in Park lakes west of the divide. A second objective was to examine whether bull trout abundance was correlated with lake trout abundance. Empirical evidence was used to test the hypothesis of Donald and Alger (1993) that lake trout, when introduced in waters with native bull trout, soon become the prevailing species.

Study Area

The five lakes surveyed in this study (Kintla, Bowman, Quartz, Logging, and McDonald) are located on the west side of the Park, in the North Fork Flathead and Middle Fork Flathead river drainages (Fig. 1). These are the largest (360-2763 ha) and deepest (60-142 m maximum depth) lakes on the west side of the Park (Table 1). They are classified as oligotrophic mountain lakes, occupying narrow glaciated mountain valleys at 961-1396 m elevation. Each lake is approximately 6-15 km long and 1- 3 km wide. Headwaters originate in snow fields of the Livingston Range at elevations extending to approximately 3000 m. The shoreline and substrate of all five lakes consists primarily of glacial rubble, dominated by cobble and large boulders, with an abundance of large woody debris along the shoreline. Each of the lakes has an alluvial fan at the upstream end where the primary inlet stream enters the lake. The littoral zone is generally steep with deltas formed where tributaries enter or landslides contact the lake.

Inlet streams to each lake are sufficiently large to provide potential spawning and rearing habitat for bull trout, though natural barriers block portions of each watershed. Morton (1968a) summarized the findings of a number of earlier investigators and noted that Park lakes provided differing potential for spawning and recruitment of bull trout and westslope cutthroat trout. Although not well documented, the highest quality spawning and rearing habitat for bull trout is believed to occur in Bowman, Quartz and Logging creeks, with more limited potential in McDonald and Kintla creeks. Due to the steep and glaciated valleys there are very few permanent lateral tributaries to the lakes and they seldom provide substantial spawning or rearing habitat for bull trout.

Lake trout were first verified in Lake

Table 1. Surface area, elevation, maximum depth, and known nonnative salmonid species composition of lakes surveyed in Glacier National Park, 2000.

Lake Name	Surface Area (ha)	Elevation (m)	Maximum Depth (m)	Nonnative Salmonids Present	Year Lake Trout Verified as Present
Kintla	688	1222	119	Lake Trout Kokanee	1962
Bowman	691	1229	77	Lake Trout Kokanee	1962
Quartz	360	1346	83	None	Not Present
Logging	444	1162	60	Lake Trout Kokanee	1984
McDonald	2763	961	142	Lake Trout Brook Trout Rainbow Trout Kokanee Lake Whitefish	1959

McDonald (1959), followed by Bowman and Kintla lakes (1962), and Logging Lake (1984) (Glacier National Park, unpublished data). Lake trout have not been found in Quartz Lake.

Recent and historical fish distribution data indicates that the outlet streams of large lakes in the Flathead drainage are seldom occupied by juvenile or adult bull trout (MBTSG 1995). Because these lakes are large and deep and they stratify, lake surface temperature greatly influences water temperatures in their outlet streams. While the streams are cold in spring and early summer, they are relatively warm later in the summer and fall with daily maxima often exceeding 15 °C. At that time, temperatures are warmer than the range preferred by bull trout for spawning and rearing (USDI Fish and Wildlife Service 1998) and bull trout spawning and rearing has not been reported in lower Kintla, Bowman, Logging, Quartz, or McDonald creeks below the lakes.

Methods

In June and July 1969, Park staff conducted a systematic gill net survey in the five study lakes to assess the survival of stocked hatchery cutthroat trout. A total of 53 nets were set overnight in the five lakes (6-15/lake). I was unable to determine depth and location of net sets or other individual net catch information since the original data sheets could not be located. A summary report described the aggregate catch of fish by species and weight in each lake (Glacier National Park, unpublished data). Specific net design was not detailed in the summary report, but the wide distribution of species and size ranges of fish captured indicate that panels of variable mesh sizes, i.e., experimental, were used.

In 1977 the U.S. Fish and Wildlife Service conducted a series of baseline limnological and fishery surveys in Park waters. Four of the five study lakes, with the exception of Quartz Lake, were surveyed. Between 8 and 18 overnight sets of 76-m (250-foot) or 91-m (300-foot) sinking gill nets were made in each lake (62 nets total), between mid-August and mid-September. Standard 76-m nets constructed with five experimental mesh sizes (19, 25, 32, 38, and 51 mm; or 3/4 to 2 inch bar measure) were used. Each mesh panel was

15 m (50 feet) long and 2 m (6 feet) deep with the panel of smallest mesh on one end, progressing to the panel of largest mesh on the opposite end of the net.

Twenty-four of 62 total net sets made in 1977 used 91-m nets that included an extra 15-m panel of 102-mm (4-inch) bar mesh. Comments in the summary report did not indicate any variation in catch efficiency in nets with the extra panel of large mesh. The author's past experience in gill netting Flathead Lake and other waters with a similar mixed species assemblage indicated that large (102-mm) mesh is usually inefficient in capturing all but the largest fish. Catch/net in the 91-m nets (~16 fish/net) was actually lower than in the standard 76-m nets (~23 fish/net), and no adjustment was made in the data to reflect the variation in net length.

In 1977 nets in each lake were set perpendicular to shore at representative sites in the upper, midsection, and lower end of each lake to incorporate a diversity of depth and habitat types. Net set duration was similar for all sets, and summaries of catch information were reported (USDI Fish and Wildlife Service 1978). Individual net catches and depths of sets were not reported, and original data sheets could not be located. Baseline surveys using the protocol developed in 1977 were not repeated in subsequent years, until 2000.

In 2000 I made an effort to duplicate the basic procedures of the 1977 survey, using sinking gill nets set overnight in the same general areas of each lake at the same time of year (14 Aug-19 Sep). The nets used were constructed of multifilament nylon and were 38 m long by 2 m deep with five panels of 19-, 25-, 32-, 38-, and 51-mm bar mesh. This has become the standard net used for bull trout surveys in other waters of the Flathead River basin (Deleray et al. 1999).

To mimic the 1977 protocol of 76-m nets, the 38-m nets deployed in 2000 in Kintla, Bowman and McDonald lakes were set in pairs, tied end to end, with the small mesh nearest shore. Total surface area of each mesh size in a pair of 38-m nets is identical to a single 76-m net, but the panels are half as long. For comparison purposes, catch in paired 38-m nets was treated as if they were from a single 76-m net. In 2000 I also surveyed Quartz Lake, which was previously surveyed in 1969 but not in 1977. I set nets in Quartz and Logging lakes singly rather than in pairs to reduce the possibility of having them snag on the abundant downed logs where they would be difficult to retrieve from a canoe.

Intensity of net sampling in 2000 was approximately half of 1977 levels (one night set at each site instead of two) to minimize mortality of bull trout. I estimated depths of each set by measuring the vertical length of line attached to floats on either end. The shallow end (small mesh size) was typically set at 3-9 m and the deep end at 9-30 m, depending on basin morphology at each site. All nets were set overnight (average 16.5 hours). Set time increased in more remote lakes due to logistic concerns, and later in the fall as day length decreased. I did not standardize net catch by hours set since that information was not available in 1969 and 1977. Set time was not considered to be an important factor since species composition and not catch per unit effort was the primary variable I evaluated.

Because the raw data for individual net catches from 1969 and 1977 were not available, only the total or average net catch for each year could be used as count data. Upon examination of preliminary results, I concluded that the number of samples (average counts) were too low to conduct meaningful statistical tests.

I identified and measured all captured fish for total length (nearest mm). I standardized 2000 net surveys to catch/76-m equivalents in order to compare with 1969 and 1977 efforts.

Results

I captured 1437 fish in the five study lakes during 2000 (Table 2) with the dominant species being mountain whitefish, longnose suckers, and lake trout. Lake trout were captured in four of the five waters surveyed, absent only in Quartz Lake. Bull

Table 2. Number of each species captured in gill nets set in Glacier National Park lakes, 2000. Number of nets has been standardized to 76-m equivalents.

Lake (No. Nets)	Bull Trout	Lake Trout	Cutthroat Trout	Mountain White-fish	Pygmy White-fish	Lake White-fish	Long-nose Sucker	Large-scale Sucker	Pea-mouth	Northern Pike-	Redside Shiner minnow
	Species										
Kintla (10)	2	45	2	187			79	25	66		
Bowman (10)	10	57		320			51	4			
Quartz (3)	20		6	85			32	2			
Logging (4)	7	12	13	112			21	2		37	
McDonald (10)	7	24		37	4	48	25	4	26	60	4

trout were captured in all five lakes, making up 0.5-3.4 percent of the catch in the four lakes where lake trout were present, but 13.8 percent of the catch in Quartz Lake. In each of the four lakes with both bull trout and lake trout present, lake trout catch exceeded that of bull trout. Lake trout comprised 5.9-12.9 percent of the total fish catch in those four lakes.

Mountain whitefish were the most ubiquitous of all species captured in 2000. They were found in all lakes and numerically dominated the catch in most waters (Table 2). Lake whitefish (*Coregonus clupeaformis*), an introduced species, were captured only in Lake McDonald. Lake whitefish were reportedly established in Lake McDonald prior to 1941 (Morton 1968b).

The 1969 survey of Kintla Lake captured 54 bull trout, comprising 94% of the balance between the two *Salvelinus* species, and 3 lake trout (6%) in 9 nets (Fig. 2). In 1977 total catch was 12 bull trout (40%) and 18 lake trout (60%) in 18 nets. In 2000 I caught 45 lake trout (96%) and only 2 bull trout (4%) in 10 net sets.

Lake trout were first reported in the voluntary creel survey from Bowman Lake in 1962 and then were documented annually in low numbers in the angler catch for several more years in the 1960s (Morton 1968a). There were no lake trout captured in either the 1969 or 1977 net surveys (Fig. 2), apparently because numbers were below detectable levels. In 1969 a total of 97 bull trout were captured in 11 nets, and in 1977 a total of 41 bull trout were captured in 18 nets in Bowman Lake. However, by 2000 lake trout had become the dominant char species in Bowman Lake. In 2000 I caught 57 lake trout (85%) and 10 bull trout (15%) in 10 sets.

In Logging Lake bull trout were captured in net surveys in 1969 (61 in 12 nets) and 1977 (6 in 10 nets), but lake trout were not detected (Fig. 2). National Park Service staff first verified the presence of lake trout in Logging Lake in 1984. My 2000 survey indicated that lake trout are now the dominant char species in this lake. I captured 12 lake trout (63%) and 7 bull trout (37%) in 4 nets.

Lake McDonald was the first lake in the Park where nonnative lake trout were verified (1959) and lake trout were frequently noted in the angler catch in the 1960s (Glacier National Park, unpublished data). Lake McDonald is geographically the closest lake to Flathead Lake within the Park (Fig. 1) and is about 100 km upstream from Flathead lake. A Park file memo, dated 1964 (Glacier National Park, unpublished), noted unusual catches of three to sixteen pound lake trout in the lower end and outlet of Lake McDonald during the last 10 days

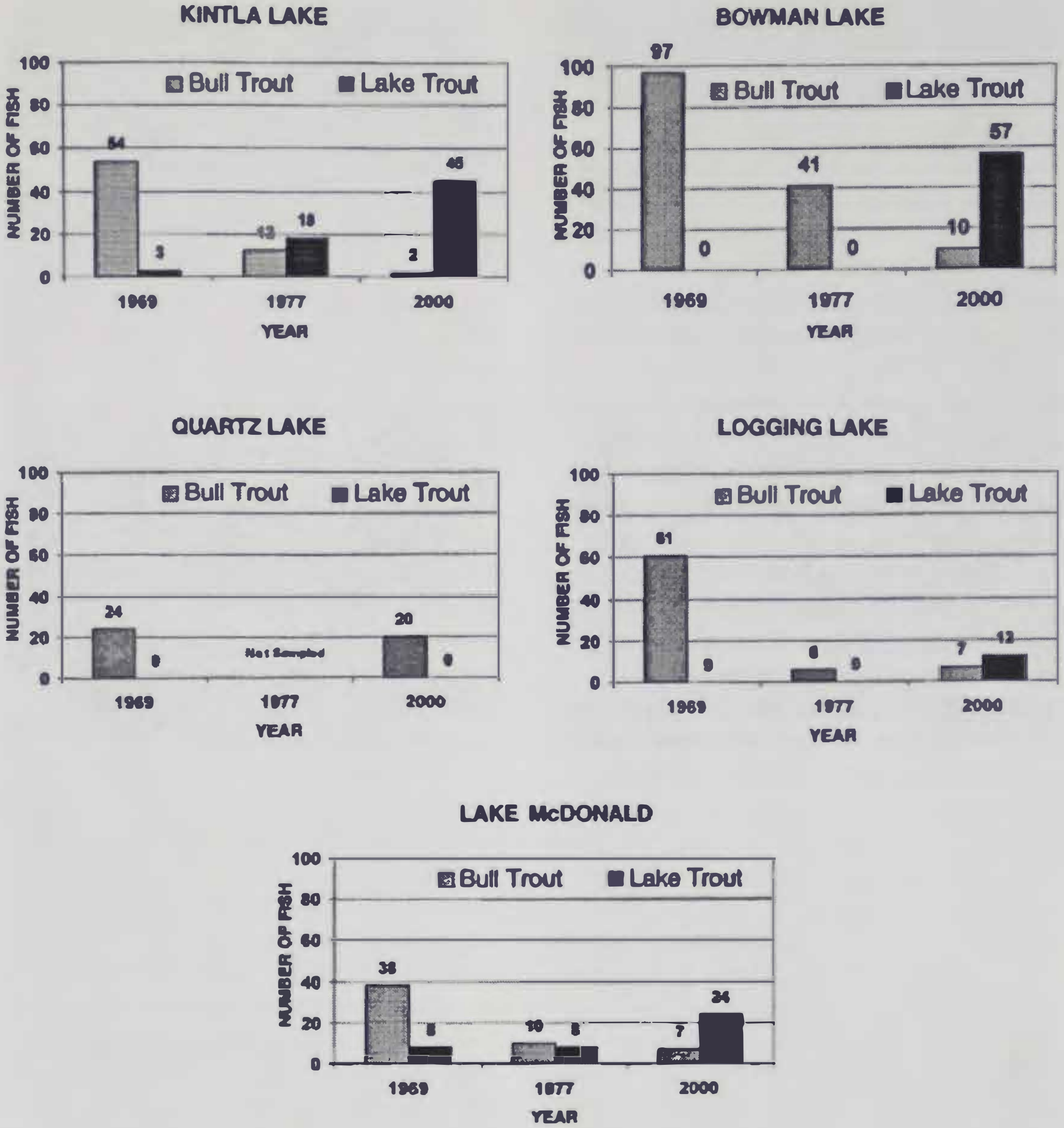

Figure 2. Comparative catch of bull trout and lake trout from gill net surveys conducted in five Glacier National Park lakes in 1969, 1977, and 2000.

of June, 1964, following a 100-year flood. The 1969 gill net survey captured 38 bull trout (83%) and 8 lake trout (17%) in 15 nets (Fig. 2). In 1977 net surveys captured 10 bull trout (56%) and 8 lake trout (44%) in 19 nets. In 2000 surveys I captured 7 bull trout (23%) and 24 lake trout (77%) in 10 nets.

In 1969, six gill net sets captured 24 bull trout in Quartz Lake. Quartz Lake was not surveyed in 1977. In the 2000 survey, I captured 20 bull trout in only three sets.

Dicussion

Bull trout populations have declined in many lakes throughout the upper Columbia River basin (USDI Fish and Wildlife Service 1998). Habitat and water quality degradation and fragmentation, past fisheries management practices and overfishing, and competition from introduced nonnative fish species are listed as three primary causes of widespread bull trout population declines (MBTSG 1995, Donald and Stelfox 1997, USDI Fish and Wildlife Service 1998).

As evidence of the effects of nonnative species, Donald and Alger (1993) evaluated the relative status of bull trout and lake trout populations in 34 lakes, where distribution of the two species overlaps in the Rocky Mountains, including portions of northwest Montana. They concluded that lake trout are usually dominant over bull trout when both species are present (either naturally or by introduction) in lakes at an elevation <1500 m. In circumstances in which lake trout are introduced into waters containing native bull trout, Donald and Alger (1993) reported that lacustrine populations of bull trout usually cannot be maintained.

Data I collected from the 2000 survey of Park lakes (Fig. 2) corroborates Donald and Alger's (1993) conclusions. Overall, comparison of the three data sets (1969, 1977, and 2000) in the five lakes indicated a broad decline in bull trout numbers and a corresponding increase in lake trout populations in Kintla, Bowman, Logging, and McDonald lakes. In Quartz Lake, where lake trout have not been found, bull trout catch appeared similar in 1969 and 2000, inferring relative stability in this population.

I recognize the limitations of drawing steadfast conclusions from these few discrete sampling points. Interpretation is further complicated by the fact that I was unable to locate raw data for the 1969 and 1977 samples, and thus could not perform statistical analysis. There also are some acknowledged inconsistencies in timing and sampling methodology. However, magnitude and direction of these changes is compelling. It strongly infers that nearly complete shifts in *Salvelinus* species composition have taken place. Furthermore, these changes have occurred independently in four separate lakes where water and habitat quality are generally not impaired and overfishing is not an issue. I conclude that bull trout abundance in four of the five Park lakes I studied has declined, probably due to interaction with nonnative lake trout, and a corresponding increase in lake trout has occurred.

Another intensively-studied bull trout population is located downstream from the Park, in Flathead Lake (Fraley and Shepard 1989). Donald and Alger (1993) noted that Flathead Lake was one of only two exceptions to their general hypothesis that lacustrine bull trout populations usually decline or are extirpated if lake trout are introduced. More recent data (MBTSG 1995, Deleray et al. 1999) indicate that, in fact, the balance of species in Flathead Lake has shifted dramatically over the past two decades. In 1981 and 1983, spring gill net series in Flathead Lake, using standard sinking nets, as in this study, caught 1.6-2.6 bull trout and 0.0-0.1 lake trout/net. During 1992-1998 annual spring gill net monitoring series captured 0.0-0.5 bull trout and 1.2-3.1 lake trout/net (Deleray et al. 1999), indicating that lake trout now heavily dominate the sympatric char species complex in Flathead Lake.

In mountain lakes of the Rocky Mountains conversion of unique native bull trout ecosystems to lake trout-dominated systems appears to be a common result once lake trout are established (see Donald and Alger 1993, Donald and Stelfox 1997). It is clear from my study that even when habitat conditions remain relatively unaltered the transition to a fish community where lake trout are the dominant piscivore may take place rapidly (Fig. 2). On an ecological scale, 20 or 30 years is a very rapid transition, given that the native fish complexes presumably have been intact for thousands of years.

Whether the introduction of lake trout will ultimately result in the complete extirpation of bull trout from the lakes I studied remains unclear. This will likely depend on several factors. Primary modifiers in the rate and extent of replacement of bull trout by lake trout will likely include migration potential of both species, extent and quality of the upstream (and in some cases downstream) spawning and rearing habitat, and structure and complexity of the food web. Ultimately, the risk of complete extirpation of bull trout may vary from system to system.

Precautions should be taken to prevent further invasions or introductions of lake trout into bull trout waters. I contend that effective recovery actions for adfluvial bull trout populations in mountain lakes, where nonnative lake trout have become established, must be directed at reducing species interaction through targeted control actions on lake trout. In former bull trout strongholds, where lake trout have become well-established, research that explores potential methods of controlling or eliminating lake trout should be a high priority. I conclude that four of the five populations of bull trout in Glacier National Park lakes that I studied are currently at high risk of extirpation, primarily due to invasion and establishment of lake trout.

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Status Of Westslope Cutthroat Trout In The Madison River Basin: Influence Of Dispersal Barriers And Stream Temperature

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Abstract

We evaluated the contemporary distribution and abundance of westslope cutthroat trout (*Oncorhynchus clarki lewisi*; WCT) in the Madison River basin, southwest Montana in relation to fish dispersal barriers and stream temperatures. Westslope cutthroat trout distribution boundaries were primarily shaped by natural fish dispersal barriers that excluded nonnative salmonids from upstream reaches. Most WCT populations occupied relatively short stream lengths ($\bar{x}$= 4.51 km, SE = 1.1), and densities ($\bar{x}$=21.9 fish >75 mm total length/100 m of stream, SE= 3.2) were generally much lower than in other drainages inside their range within Montana. Where WCT and nonnative salmonids segregated without the influence of dispersal barriers, distribution boundaries were related to stream temperature with WCT occupying colder stream reaches. Patterns of fish occurrence and stream temperature indicated that WCT have been displaced from warmer stream habitats and now occupy a narrower and colder range of stream temperatures than they did historically. Isolated populations of WCT encountered a higher and greater range of average summer stream temperatures and reached higher abundances than those populations in streams without dispersal barriers. This suggests that while colder stream temperatures may provide a competitive advantage for WCT relative to nonnative species, these habitats may be marginal due to lower individual fitness and reproductive success of WCT. Because low population sizes and isolation place many WCT populations at risk of extirpation, we recommend that WCT populations in the Madison Basin be replicated and expanded downstream to ensure their long term persistence.

Key words: competition, dispersal barrier, fish distribution, hybridization, *Oncorhynchus clarki, O. c. lewisi*, stream temperature, westslope cutthroat trout

Introduction

Because of their popularity as sport fish, many salmonid species have been transplanted outside their native ranges throughout North America. Introductions of nonnative salmonids have typically resulted in range constriction or elimination of native species as a result of predation, competition, or hybridization (Gresswell 1988, Behnke 1992). In some situations native salmonids have persisted in the presence of introduced species, but the mechanisms that regulate displacement, and the habitat conditions that provide refuges for native species are not well understood (Fausch 1988, Gresswell 1988, Bozek and Hubert 1992).

As with other interior stocks of cutthroat trout (*Oncorhynchus clarki*), populations of westslope cutthroat trout (*O. c. lewisi*; WCT) have declined throughout their historical range (Hanzel 1959, Liknes and Graham 1988, Behnke 1992). In

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Montana, declines of WCT have been most substantial within the Missouri River basin, with genetically pure populations occupying <5 percent of the historical range (Shepard et al. 1997). The original distribution of WCT within the Missouri River basin is thought to include the entire Missouri River drainage upstream from Fort Benton, Montana, including the Gallatin, Madison, and Jefferson drainages, as well as the headwaters of the Judith, Milk, and Marias rivers, which join the Missouri River downstream from Fort Benton (Behnke 1992). Prior to about 1900, the Madison River and its principal tributaries supported abundant populations of WCT upstream to barrier falls on the lower Firehole and Gibbon rivers in Yellowstone National Park (Jordan 1891). WCT abundance and distribution declined rapidly early in the 1900s (USDI Fish And Wildlife Service 1999) and by the early 1950's WCT no longer occurred in the Madison River or its principal tributaries within Yellowstone National Park (Benson et al. 1959), and were restricted to headwater habitats elsewhere in the drainage (Hanzel 1959).

Factors responsible for declines of WCT include habitat alterations caused by land and water use practices, overharvest, and introductions of nonnative fishes (Hanzel 1959; Liknes and Graham 1988, Behnke 1992, McIntyre and Rieman 1995). Interactions with nonnative species through predation, competition, or hybridization probably constitute the greatest contemporary factor responsible for the loss of WCT populations (Allendorf and Leary 1988, Liknes and Graham 1988, USDI Fish And Wildlife Service 1999).

Extant populations of WCT within the Madison River drainage are now restricted to headwater habitats, often above the upstream limit of nonnative salmonids (Sloat et al. 2000). In allopatry, WCT are capable of inhabiting a much broader range of habitats. Historically, WCT occupied small headwater streams and larger rivers, as well as mid- to low-elevation lakes (Shepard et al. 1984, Marnel 1988, Behnke 1992) and individuals are known to make extensive migrations between these habitats (Bjornn and Mallet 1964, Shepard et al. 1984, Schmetterling 2001). Interactive niche compression resulting from the influence of nonnative salmonids may partially explain the confinement of WCT to headwater habitats (Mullan et al. 1992). Fausch (1989) hypothesized that colder, higher gradient headwater habitats provide refuges for cutthroat trout, where nonnative salmonids either cannot persist or where environmental conditions tip the balance of interspecific competition to favor cutthroat trout. Behnke (1992) suggested that cutthroat trout might have a selective advantage over nonnative trout in headwater areas because they may function better in cold environments. Field and laboratory studies have demonstrated the importance of temperature in shaping cutthroat trout distribution (DeStaso and Rahel 1994, Mullan et al. 1992, Dunham et al. 1999). Cutthroat trout also have slightly lower thermal tolerances than nonnative salmonids (Heath 1963, Feldmuth and Erikson 1978, DeStaso and Rahel 1994). Even relatively small differences in salmonid thermal tolerances can reflect substantial differences in growth optima (Takimi et al. 1997), competitive ability (DeStaso and Rahel 1994), and regional distributions (Fausch et al. 1994). Consequently, the influence of temperature on the distribution of WCT has become a central concern in management for this subspecies. However, despite evidence that temperature is important, relatively little information is available to assess thermal regimes that provide suitable habitat for WCT or provide refuges from competition and hybridization with introduced salmonids.

Another factor potentially affecting the distribution of WCT in streams is the occurrence of dispersal barriers. Natural and anthropogenic dispersal barriers may restrict the distribution of salmonids (Kruse et al. 1997, Dunham et al. 1999) and in some cases protect native salmonids from potential displacement by nonnative species (Rinne and Turner 1991, Young et al. 1996).

Our goal was to explore how spatial patterns of fish dispersal barriers and stream temperature influenced the distribution of remnant WCT populations in the Madison River basin, Montana. Our specific objectives were to: 1) describe the contemporary distribution and abundance of WCT in the Madison River basin; 2) determine the influence of fish dispersal barriers on the distribution and abundance of WCT; and 3) determine the thermal characteristics of habitats occupied and unoccupied by WCT.

METHODS

Study Area

This study was conducted in the 906-km² Madison River Valley, a north-trending intermontane basin located in southwest Montana. The Madison River is formed at the confluence of the Firehole and Gibbon rivers in Yellowstone National Park and flows approximately 195 km northward before joining the Gallatin and Jefferson rivers to form the Missouri River near the town of Three Forks, Montana. Our study focused on tributaries to the 101 km section of the Madison River between Hebgen and Ennis reservoirs (Fig 1).

The study area was bordered by mountain ranges that differ in their morphology. The Madison Range forms the eastern boundary of the study area and rises sharply from the valley floor to peak elevations exceeding 3200 m. The Gravelly Range forms the western border of the study area and is less rugged than the Madison Range with elevations not exceeding 2900 m. Although the alluvial plain in the Madison River valley is predominantly in private ownership, the majority of the basin is public and managed by the USDA Forest Service (FS). The primary land use in the Madison Valley is livestock grazing with localized dryland and irrigated agriculture. Limited logging has occurred on FS land in the Gravelly Mountains. Land use is restricted in the Lee Metcalf Wilderness Area, which encompasses most of the Madison Range within our study area. Snowmelt drives flow regimes in tributary streams, and peak discharges occurred in May and June. Streams ranged from first to fourth-order (measured from 1:24,000-scale USGS topographic maps after Strahler [1957]) with drainage areas between 9.2 and 128.8 km². Mixed conifer stands dominated riparian vegetation adjacent to study streams in headwater reaches, except along unconstrained reaches where willows (*Salix* spp.) dominated. Willows, sedges (*Carex* spp.), and grasses typically dominated downstream reaches. Sloat et al. (2000) provide detailed descriptions of individual study streams.

The climate of the Madison River Valley is typical of high-elevation intermontane basins with mild summers and cold winters. The average annual precipitation is 33.7 cm, and the average annual air temperature is 6.4 °C on the valley floor (NOAA 1999).

During the last 100 years, several nonnative salmonid species have been introduced into the Madison River (Montana Fish, Wildlife and Parks 2000). Rainbow trout (*O. mykiss*) and brown trout (*Salmo trutta*) were stocked periodically into the Madison River and its tributaries as early as 1889 (USDI Fish And Wildlife Service 1999) and were well established by the 1930's (USDI Fish And Wildlife Service 1954). Releases of hatchery-raised rainbow trout into the Madison River continued until 1974 (Vincent 1987). Yellowstone cutthroat trout (*O. c. bouvieri*) have been stocked in the Madison River drainage since the early 1950's, primarily in high mountain lakes, but also in many streams. Yellowstone cutthroat trout stocking in the Madison Range continued through the period of our study (Montana Fish, Wildlife and Parks 2000). Within the Madison River drainage the only native salmonids to co-occur with WCT were mountain whitefish (*Prosopium williamsoni*) and arctic grayling (*Thymallus arcticus*). The Madison River grayling population disappeared as early as 1920 (USDI Fish And Wildlife Service 1954), and only a vestigial population now inhabits Ennis Reservoir. Native nonsalmonid

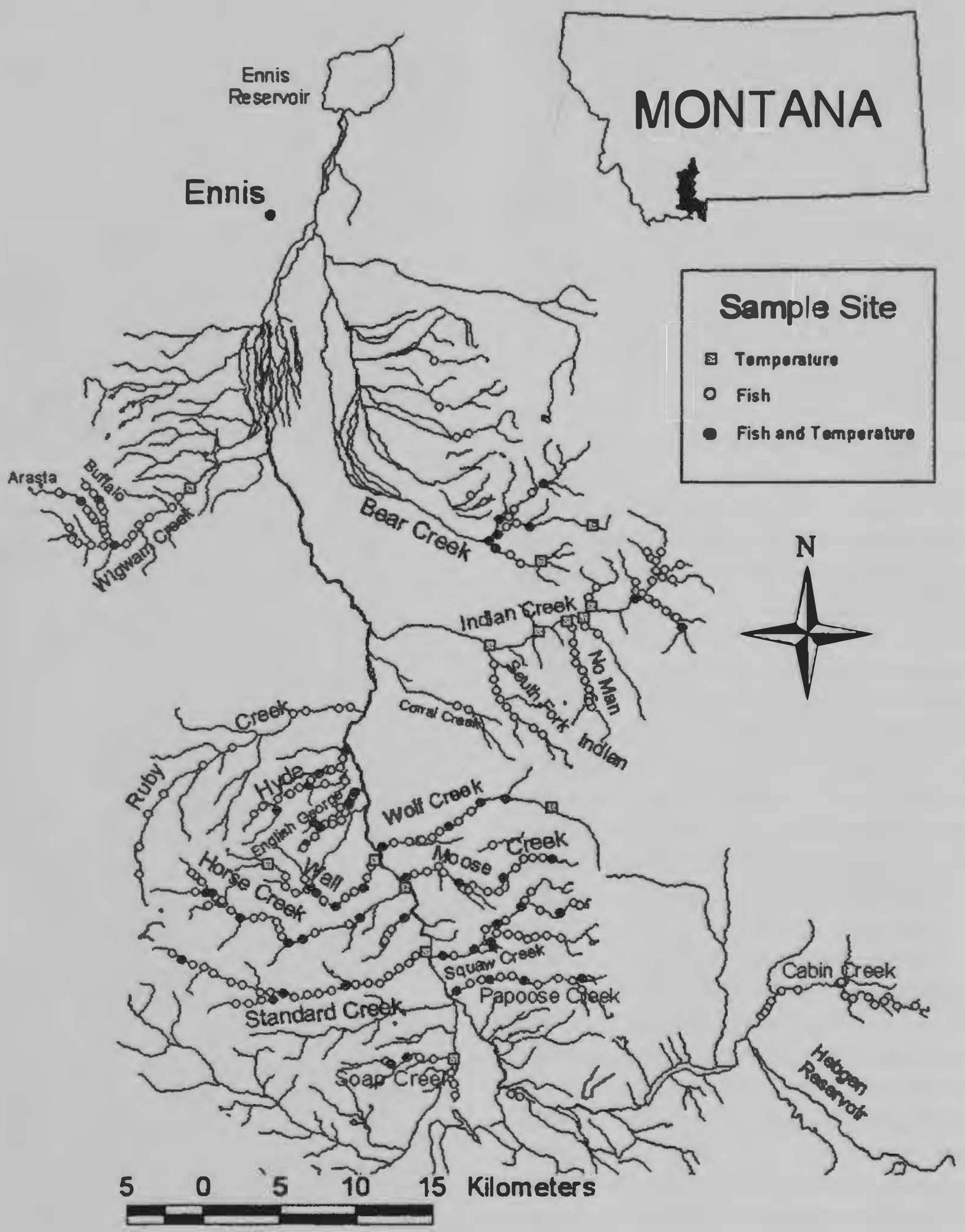

Figure 1. Map of Madison River drainage from Hebgen Reservoir to Ennis, Montana showing names of major streams sampled, and sample sites by type (Temperature = temperature recording site; Fish = electrofishing sample sites; Fish and Temperature = temperature recording and electrofishing sample site). Lower reaches of Bear, Corral, Indian, and Wigwam creeks were dry.

fishes present in the Madison River drainage include white sucker (*Catostomus commersoni*), longnose sucker (*C. catostomus*), mountain sucker (*C. platyrhynchus*), longnose dace (*Rhinichthys cataractae*), and mottled sculpin (*Cottus bairdi*) (Federal Energy Regulatory Commission 1997) but mottled sculpin was the only nonsalmonid species observed in Madison River tributaries.

Fish Distribution Sampling

We employed a systematic sampling

design to determine relative abundance and distribution of fishes. Our primary objective was to locate remnant WCT populations and we did not sample streams if previous inventories indicated that they contained only nonnative species. We sampled streams at 0.8 km (0.5 mi) intervals by single-pass electrofishing and at 3.2 km (2.0 mi) intervals, multiple-pass depletion population estimates were made (VanDeventer and Platts 1985). Smith-Root electrofishers (Models SR-15B, SR-12B) were used for all electrofishing. We slightly modified this protocol in some streams with more frequent sampling to document the upper and lower extent of distribution of each fish species. To help ensure that we captured all species present, sample section lengths were at least 35 times the average wetted stream width (Lyons 1992). Sample sites were referenced by stream kilometer starting at the mouth and by latitude and longitude obtained from a Global Positioning System (GPS). Sampling progressed upstream until trout were no longer present; then an additional upstream site was usually sampled to ensure fish absence. We recorded total length (TL) and weight for all captured salmonids. Most fish distributions were sampled during the summers of 1997, 1998, and 1999, but data for Soap Creek were collected in 1995.

When conducting multiple depletion population estimates, if field-calculated probabilities of capture (calculated as 1-[C2/C1]; where C1= number captured on the first pass, and C2 = number captured on second pass) were <0.80 after two passes, up to two additional passes were made until capture probabilities reached 0.80 (cf., Riley and Fausch 1992). Relative abundance was calculated by species as the number of fish (≥75 mm TL)/100 m of stream captured in the first (or only) electrofishing pass. Population estimates were calculated using a maximum likelihood estimator within the MICROFISH program (Van Deventer and Platts 1985) and standardized as the number of fish/100 m of stream length.

Field identification of fish species was based on spotting pattern, body color, and presence/absence of an orange "cutthroat" slash below the lower mandible described for WCT in Behnke (1992) and was confirmed with genetic testing for most streams. Either whole fish or, most often, fin samples from fish identified as WCT were taken for genetic analysis. Genetic characteristics were determined by horizontal starch gel electrophoresis (whole fish) or by Paired Interspersed Nuclear DNA Element-PCR (PINE [fin clips]) by the University of Montana Wild Trout and Salmon Genetics Laboratory. Where possible, we sampled 25 field-identified WCT/stream, which provides a 95-percent chance of detecting as little as a 1 percent Yellowstone cutthroat or rainbow trout genetic contribution to a hybridized population of WCT (Spruell and Miller 1999). Often, however, the sample size was lower than 25 fish (Appendix A). Where possible, a portion of the 25-fish sample was captured at each of multiple sampling sites within a stream to test for longitudinal changes in genetic composition within a population. Fish were considered WCT if frequencies of alleles characteristic of WCT were ≥90 percent. This was based on management guidelines of Montana Fish, Wildlife and Parks that provide populations with >90 percent genetic purity the same protections afforded pure WCT because these populations indicate suitable habitat for WCT and may have genetic value for future conservation efforts (Montana Fish, Wildlife and Parks 1999). Hybridized populations of WCT with >10 percent introgression were classified as nonnative salmonids.

Potential barriers to fish movement, defined as structures with vertical drops at least 1.5 m high (Stuber et al. 1988, Kruse et al. 1997), were identified by surveying the entire length of each tributary. Barrier locations were referenced by latitude and longitude using a hand-held GPS unit and input into the geographic information system (GIS) computer program ArcView (Environmental Systems Research Institute 1999) and projected on 1:100,000 stream hydrography layers. Barriers consisted of waterfalls, decadent beaver (*Castor canadensis*) dam complexes, and irrigation diversion dams.

We derived length of habitat occupied by WCT and nonnative salmonids for each tributary drainage in ArcView using 1:100,000 stream hydrography layers and then made comparisons using Welch's modified t-test, which does not assume equality of group variances (Zar 1984). Occupied habitat lengths were defined as the total occupied stream kilometers in a drainage not interrupted by a dispersal barrier, and did not include the main stem of the Madison River.

Stream Temperature Sampling

Continuously recording digital thermographs ("Hobo" and "Stowaway" models, Onset Corp.; http://www.onsetcomp.com) were used to record water temperatures in first- to fourth-order streams (Strahler 1957) across the Madison River basin (Fig. 1). Thermographs were capable of measuring temperatures ranging from -5 to 37 °C with an accuracy of ± 0.2 °C. Prior to field deployment, thermographs were calibrated against a National Institute of Science and Technology hand held thermometer at 3, 9, and 20 °C.

Thermographs were deployed from early July to late September. Where trout distributions were known a priori, we placed thermographs at upper and lower distribution boundaries. For many streams where distributions were not known, thermographs were uniformly distributed along the stream's length. We placed a minimum of three thermographs from 1 to 7 km apart in principal study streams and at the mouths of smaller tributaries. Thermographs were placed in well-mixed run or pool habitats and were shielded from direct sunlight. Thermographs recorded hourly stream temperatures that we summarized into daily maxima, minima, and means.

Because of a limited number of thermographs available for this study and the extensive time involved in placing thermographs in the field, not all tributaries sampled for fish had thermographs. We collected stream temperature data from 71 sites in 1999 but also measured stream temperatures at six sites during 1998 and two sites in 1997.

The following temperature metrics were calculated for sites where both fish abundance and stream temperature data were collected: Maximum Average Daily Temperature (MDAT)—the maximum of all average daily temperatures within a year; Maximum Daily Maximum Temperature (MDMT)—the maximum of all maximum daily temperatures within a year; Maximum Weekly Average Temperature (MWAT)—the maximum seven-day average of daily average water temperatures; Maximum Weekly Maximum Temperature (MWMT)—the maximum seven-day average of daily maximum water temperatures; and Degree Days (DD): the sum of average daily temperatures over 0 °C. During 1999, not all thermographs were in place by 1 July. Therefore, to facilitate comparison, degree-days were calculated from 8 July to 15 September when all thermographs were in place and recording. We used t-tests and analysis of variance (ANOVA) ($\alpha = 0.05$) to test the hypothesis that stream temperatures were significantly colder at sites occupied by WCT than those occupied only by nonnative salmonids. We did not include sites if fish densities were <3 fish >75 mm TL/100 m of stream because extremely low densities of fish, indicating potential habitat limitations at some locations, may have biased the analysis. Fish sampling events were matched with temperature records corresponding most closely in time, but in some cases stream temperatures were not measured during the same year as the fish sampling event. For this analysis, we made two assumptions: 1) fish distribution boundaries did not change over the relatively short time period of this study; and 2) measured stream temperatures were representative of temperatures experienced by fish during the year fish distribution data were collected.

Results

Fish Distribution

We determined the distribution of WCT in the Madison River drainage between Hebgen and Ennis reservoirs using samples from 318 locations in 58 streams within 18 different sub-drainages (Fig. 2). Except for Trail Creek from which data were unavailable, results for genetic testing of all field-identified WCT populations appear in Appendix A. Westslope cutthroat trout (>90% purity) were present in 79 (25 %) of 318 sites sampled—portions of 17 of 58 streams. We found nonnative trout species, including rainbow, brown, and Yellowstone cutthroat trout, as well as hybridized WCT with >10 percent introgression, in 133 (42 %) of 318 sample sites. Hybrid cutthroat trout were present in 48 of the 133 sample sites occupied by nonnative salmonids (15% of all sample sites). No fish were captured in 106 sample sites (Fig. 2).

Within the Madison River basin, distribution of WCT was concentrated in streams draining the Gravelly Range. In this range, we found WCT in six of nine sampled sub-drainages. Because some sub-drainages included more than one occupied stream, these six sub-drainages represented occurrence in 11 streams. All WCT populations but one (Arasta Creek) were isolated from nonnative species by dispersal barriers. Natural barriers to fish dispersal were found in eight of the nine sub-drainages sampled in the Gravelly Range. In six sub-drainages, nonnative salmonids were present up to the base of the barrier and only WCT were present upstream. Nonnative salmonids were present both above and below barriers (2-5 m high vertical falls) to fish migration in the two remaining sub-drainages (3 streams).

Except in Hyde Creek, where a large beaver dam complex prohibited upstream migration of nonnative salmonids, all dispersal barriers in streams draining the Gravelly Range consisted of waterfalls. Typically, fish dispersal barriers were located relatively low in streams draining the Gravelly Range. The average distance above the stream's mouth and mean elevation of dispersal barriers in the Gravelly Range were 3.9 km (SE= 1.7) and 1954 m (SE = 79), respectively.

Westslope cutthroat trout were found less frequently in streams draining the Madison Range. Only 4 of 10 sub-drainages (6 of 35 streams) sampled in the Madison Range supported WCT (Fig. 2), and in contrast to the Gravelly Range, we found only one WCT population (Cabin Creek; Fig. 2) above a natural fish migration barrier. Dispersal barriers were found on 10 of 33 streams supporting fish in the Madison Range (Fig. 2). In eight of these streams, fish were present up to the base of a waterfall barrier and absent upstream. We found introduced Yellowstone cutthroat trout above waterfall barriers in No Man Creek that contained a headwater lake regularly stocked by Montana Fish, Wildlife and Parks (2000). Additionally, a very small population (<50 individuals) of WCT was isolated above an irrigation diversion dam in Trail Fork Bear Creek. Fish dispersal barriers occurred significantly farther upstream ($\bar{x}$=12.5 km, SE=2.9) and at higher elevations ($\bar{x}$=2267 m, SE= 62) than in the Gravelly Range (*t*-tests, $P<0.05$).

Basin-wide, WCT were sympatric with nonnative trout in only two sample sites, one each in Standard and Hyde creeks (Fig. 2), each located directly below barriers which protected upstream WCT populations. The presence of WCT at these two sites may represent downstream migrants and not a healthy population since only 2 and 3 individuals were captured at these sites in Hyde and Standard creeks, respectively.

Relative abundance of WCT captured during a single electrofishing pass ranged from 1 to 40 fish/100 m of stream length ($\bar{x}$= 10.8, SE= 1, n =79), compared to a range of 1 to 84 for nonnative salmonids ($\bar{x}$=9.7, SE= 1, n=133). Mean relative abundance of WCT was not significantly different from nonnative salmonid species (*t*-test, P=0.43). Multiple depletion density estimates ranged from 3 to 40 fish/100 m of

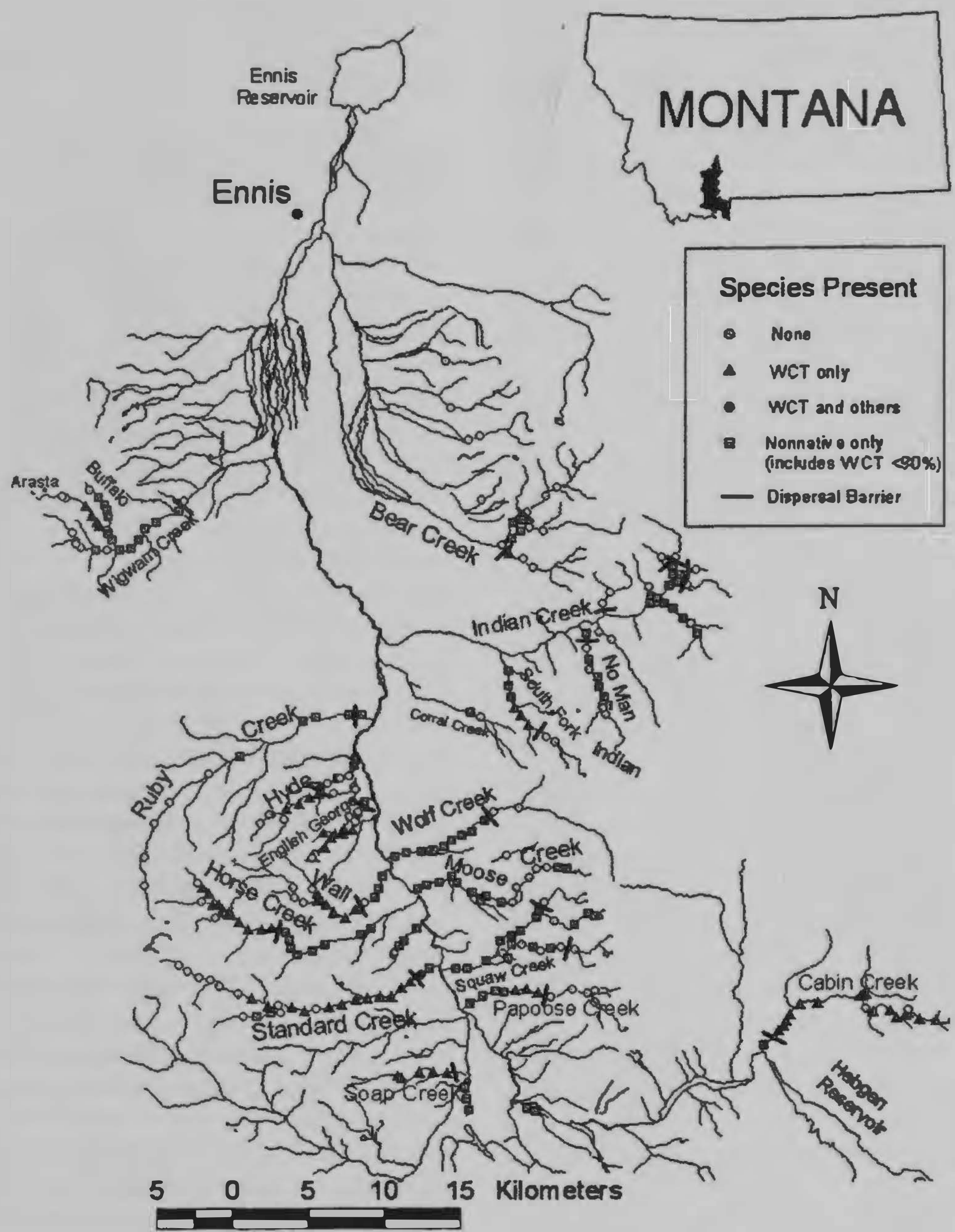

Figure 2. Map of upper Madison River drainage showing the distribution of westslope cutthroat trout with >90 percent genetic purity (WCT), nonnative salmonids and fish dispersal barriers

stream length ($\bar{x}$= 21.9, SE=3.2, n=20) for WCT, compared to 1 to 185 for nonnative salmonids ($\bar{x}$=25.6, SE=7.8, n=34). Estimated densities were not significantly different (t-test, P= 0.66). Based on capture probabilities derived from multiple depletion estimates, the efficiency of single-pass removals was approximately 80 percent for all species combined and was slightly higher for nonnative salmonids (82%) than WCT (79%), but this difference was not significant (t-test; P=0.79).

Non-isolated populations of WCT were found in only three streams: Papoose, South Fork Indian, and Arasta creeks (Fig. 2). Despite their strong association with dispersal barriers, the length of habitat occupied by WCT per sub-drainage ($\overline{x}$ = 4.51 km, SE= 1.1) did not differ from that occupied by nonnative salmonids ($\overline{x}$ = 4.99 km, SE= 1.2; *t*-test, *P*=0.77). However, when we compared occupied habitat lengths for only WCT populations, isolated WCT populations occupied longer stream lengths than did populations not isolated by fish barriers (*t*-test, *P*<0.05). Isolated WCT occupied an average stream length of 7.5 km (*n*= 8, SE= 2.2), whereas all three non-isolated populations occupied approximately 2.4 km of stream.

Westslope cutthroat trout also were more abundant (*t*-test, *P*<0.001) at sites above dispersal barriers. Mean WCT abundance at sites above physical dispersal barriers was 12.8 fish/100 m (SE=1.1) compared to 3.8 fish/100 m (SE=0.8) at sites not influenced by physical dispersal barriers. This difference in fish abundance was not made up by the presence of other species in the non-isolated cases and did not appear to be a function of limited physical habitat.

Stream Temperature

Water temperature patterns varied considerably both among and within streams. Trends in daily water temperatures at sites measured during multiple years were similar across years. Mean stream temperatures were highly correlated with daily maxima and minima. Stream temperatures fluctuated from as little as 2.3 to as much as 16.7 °C daily. Ranges of daily stream temperatures were weakly correlated with daily means but were more closely correlated with daily maxima. Average summer stream temperatures were colder in streams draining the Madison Range ($\bar{x}$ = 7.6 °C, SE=0.04) than the Gravelly Range ($\bar{x}$= 8.6 °C, SE=0.05) (*t*-test, *P*<0.001).

Westslope cutthroat trout were associated with habitats where average and maximum daily stream temperatures generally remained below 12 and 16 °C, respectively (1 Jul-15 Sep; Fig. 3). Maximum daily average temperatures (MDAT) ranged from 7.2 to 12.7 °C, and maximum daily maximum temperatures (MDMT) ranged from 9.9 to 16.5 °C at sites occupied by WCT during the summer sampling period (Table 1). Thermal regimes differed significantly between sites occupied by WCT and nonnative salmonids. Although there was considerable overlap, all stream temperature metrics tested were significantly lower at sites occupied by WCT than sites occupied solely by nonnative salmonids (Table 1).

All temperature metrics were lower for sites with WCT (*t*-tests, *P*<0.05) compared to sites occupied by rainbow trout. The distribution of rainbow trout coincided with a 1-3 °C warmer range of stream temperatures than those occupied by WCT. Rainbow trout occupied sites with maximum average daily stream temperatures between 9.2 and 13.1 °C, and maximum daily stream temperatures between 12.3 and 18.4 °C.

At the basin level, no statistical differences were found between sites occupied by rainbow x cutthroat trout hybrids and those occupied by WCT for any of the temperature metrics examined (*t*-tests, *P*>0.05). However, in at least one stream, temperature differences corresponded with distribution boundaries of WCT and nonnative species, including rainbow x cutthroat trout hybrids (Fig. 4). Westslope cutthroat trout segregated from nonnative salmonids without the influence of a dispersal barrier in Papoose Creek. In Papoose Creek, thermographs were placed at the upper distribution boundary of WCT, the upper distribution boundary of nonnative trout species, and at the stream's mouth. Average daily stream temperatures differed at all three sites (ANOVA, *P*<0.001) with average daily stream temperatures becoming progressively colder at upper stream sample sites. Average daily stream temperatures at the uppermost site in Papoose Creek were also lower than the "coldest" site where nonnative salmonids were captured (Horse Creek, km 8.8) in the Madison River drainage during this study (*t*-test, *P*< 0.001).

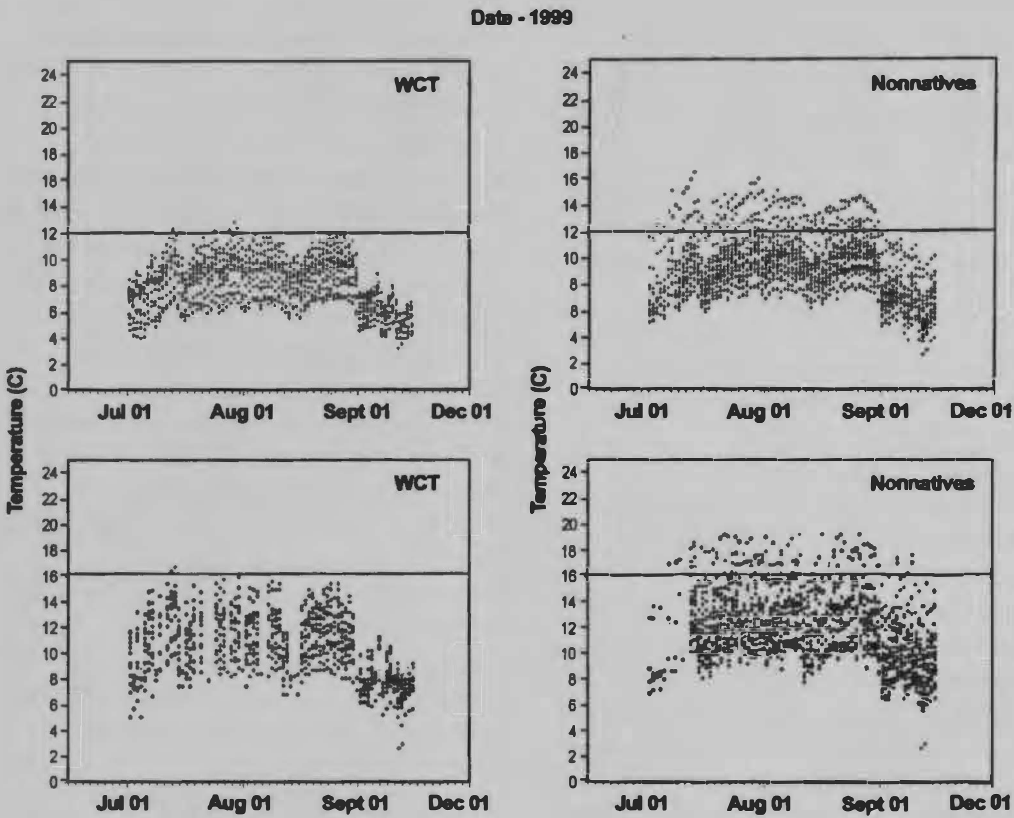

Figure 3. Average (upper) and maximum (bottom) daily stream temperatures at sites occupied by westslope cutthroat trout (WCT; n=16) and nonnative salmonids (Nonnatives; n= 25), with reference lines drawn at 12 and 16 °C, respectively

Populations of WCT above barriers encountered a greater range of average summer stream temperatures and slightly warmer stream temperatures (range= 6.6-11.8 °C, n=14) than those in streams without dispersal barriers (range= 5.9-8.4 °C, n=2). These slightly warmer thermal regimes translated into a higher number of degree days at sites above dispersal barriers ($\bar{x}$=694, SE=39) than non-isolated sites ($\bar{x}$ =564, SE= 98), but this difference was not significant (t-test, P>0.05).

Warm stream temperatures appeared to limit the lower distribution boundary of WCT in one sub-drainage. In English George and South Fork English George creeks, allopatric WCT located above a fish dispersal barrier were absent or rare (<1 fish/100 m) at sites where average daily stream temperatures warmed to 16 °C, and maximum daily stream temperatures warmed to 24 °C during the 1999 sampling season. In contrast, WCT were moderately abundant (mean abundance= 9 fish/100 m) in upstream sites where average daily water temperatures remained between 4 and 10 °C and maximum recorded stream temperatures remained below 12 °C during the summer sampling period.

Discussion

Fish Distribution

Distribution of WCT in the Madison River drainage between Hebgen and Ennis reservoirs was concentrated in streams draining the Gravelly Range (Fig. 2) and was primarily shaped by natural fish dispersal barriers that excluded nonnative salmonids from upstream reaches. We hypothesized that barriers might isolate WCT from potential hybridization or competition with nonnative salmonids. This appeared to be the case in streams draining the Gravelly Range where most perennial streams supported isolated

Table 1. Mean and range of five temperature metrics (see text for definitions) at sites occupied by westslope cutthroat trout (WCT) and sites occupied by nonnative trout species.

Temperature metric	WCT	Nonnative trout	P-*value*[a]
MDAT	9.8 (7.2-12.7)	11.1 (8.1-16.3)	0.033
MDMT	13.2 (9.9-16.5)	14.5 (10.6-22.0)	0 050
MWAT	9.5 (7.1-11.7)	10.6 (7.8-15.1)	0.022
MWMT	12.3 (9.3-15.3)	13.8 (10.0-23.1)	0.027
DD	563.9 (414.5-693.2)	626.7 (465.7-882.6)	0.030

[a] Welch's modified t-test.

populations of WCT. However, except in Cabin Creek where a geologic barrier was located relatively close to the stream mouth, WCT did not occur above natural dispersal barriers in the Madison Range. Patterns of fish occurrence indicated that the location of dispersal barriers within a stream network was important in determining the presence or absence of WCT. Although dispersal barriers were equally common in the Gravelly Range (8 of 23 streams) and the Madison Range (13 of 35 streams), fish barriers occurred significantly closer to stream mouths and at lower elevations in the Gravelly Range, which may provide insight into WCT distribution patterns in the Madison River drainage.

Isolated populations of salmonids face a variety of extinction risks through environmental and demographic variation due to limited physical space and small population sizes associated with fragmented habitats (Rieman et al. 1993). Smaller, more isolated populations are less likely to persist because 1) small populations face a higher risk of extinction through demographic and environmental stochasticity, and 2) isolated populations have no possibility of demographic support or recolonization through dispersal from surrounding populations (Rieman and McIntyre 1995, Dunham et al. 1996). Flood flows, debris torrents, drought, and fires can locally extirpate trout populations (Propst et al. 1992). If WCT naturally occurred above barriers in streams draining the Madison Range, catastrophic events may have limited their persistence in these areas. However, for many streams we do not know if WCT ever had access to reaches above dispersal barriers. In Cherry Creek, a large isolated sub-drainage outside our study area but within the Madison River drainage, native fish were absent from all of the 90 km of contiguous stream habitat above an 8-m high barrier (Bramblett 1998). Because of the large size and hydrologic complexity of this sub-drainage (Bramblett 1998), the absence of native fish species above this barrier strongly suggests that WCT were historically absent above the falls rather than extirpated due to stochastic events. In our study fishless reaches above waterfalls >10 m high in several streams within the Indian Creek sub-drainage (Fig. 2) also may represent sites that were never colonized by WCT. Consequently, it is unlikely that all fishless reaches in the Madison Range have resulted from localized population extirpations.

Local extirpations of isolated salmonid populations as a result of catastrophic events have been documented elsewhere (e.g., Propst et al. 1992). Kruse et al. (1997) found that Yellowstone cutthroat trout were absent above natural dispersal

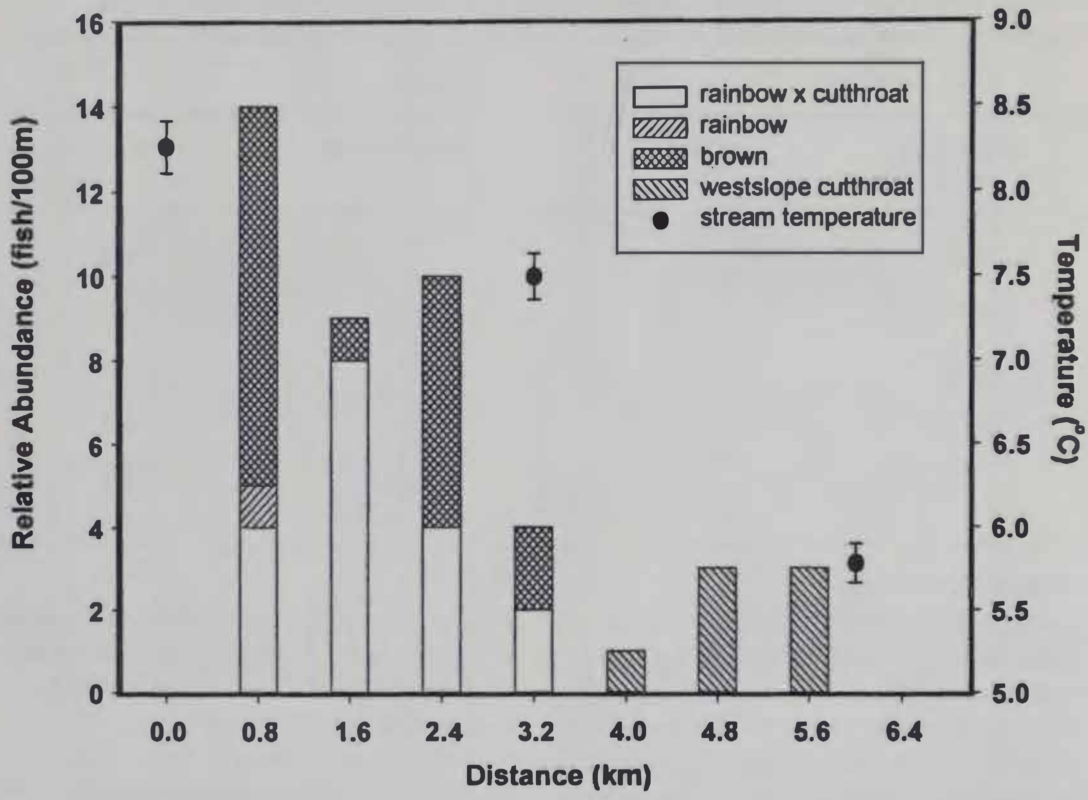

Figure 4. Relative abundance of rainbow, brown, rainbow x cutthroat hybrid, and westslope cutthroat trout greater than 75 mm (left axis) and average summer stream temperatures (right axis) in Papoose Creek by kilometer from stream mouth. Vertical lines represent standard errors.

barriers in the Wood and Greybull river drainages, Wyoming. They were unsure of historic presence, but if fish had access to these areas, they suggested that relatively short stream lengths above barriers, poor habitat conditions, and relatively common occurrences of catastrophic events could have limited their persistence. Similarly, Dunham et al. (1996) suggested that the general absence of Lahontan cutthroat trout populations above natural dispersal barriers was likely a byproduct of high extinction and low recolonization or population rescue probabilities in such small, isolated habitats. In our study, despite apparently suitable physical habitat (Sloat et al. 2000) fish were absent above a relatively recent barrier formed by a large debris jam in Wolf Creek (Fig. 2), suggesting that WCT had been eliminated from this historically accessible stream reach.

While not all fish-less stream reaches above barriers represent sites where cutthroat trout have been extirpated, this does not diminish the risk of extinction for small, geographically restricted populations. Where WCT currently exist above barriers in the Madison River drainage, low population sizes and isolation may place many of these populations at risk. Although abundances of WCT were not significantly lower than those of nonnative salmonids, relative abundances of all salmonid species in the Madison River drainage were generally much lower than in tributaries from other drainages in the upper Missouri and upper Clark Fork river basins in Montana (Sloat et al. 2000). Low abundance of trout in Madison River tributaries may be related to the relatively high elevation of this river basin, inherent geologic instability that translates into somewhat unstable stream channels, and moderate to low productivity of its watersheds (Sloat et al. 2000).

In most streams, WCT populations existed in relatively short stream reaches (mean occupied length= 4.5 km). Based on an empirical evaluation of translocation success, Harrig (2000) suggested that stream segments <5.7 km long may have insufficient space to sustain adult and juvenile greenback cutthroat trout (*O. c. stomias*). Hilderbrand and Kershner (2000) developed a simple relationship between observed cutthroat trout abundances, the proportion of individuals leaving a population through emigration and mortality, and desired population sizes to estimate the minimum stream length (MSL) necessary to maintain viable cutthroat trout populations. Following earlier work by Allendorf et al. (1997), they recommended a population benchmark of 2500 individuals >75 mm long to insure the long-term persistence of isolated populations. Based on a target population size of 2500 individuals, and assuming no proportional loss of individuals, only two streams sampled in this study have occupied MSL's that meet criteria for long term persistence presented by Hilderbrand and Kershner (2000) (Table 2).

Limited space does not necessarily mean that a population will become extinct (Hilderbrand and Kershner 2000). Some fish populations have persisted for extended periods in small habitat patches isolated by natural barriers and may have adapted to restricted space (Northcote et al. 1970, Northcote 1981, 1992). Northcote (1981) reported that heritable differences in rheotaxis between rainbow trout populations above and below a waterfall were genetically coded. Similarly, Shepard et al. (1998) found that the proportion of stream dwelling WCT moving 0.5 km or longer was negatively correlated to the level of isolation experienced by the population. While these local adaptations may be advantageous for individuals in restricted habitats, adaptations to stochastic events such as extreme floods, debris flows, or droughts may be unlikely because either the intensity or the time between such events is too great (Poff 1992). Additionally, traits that confer the greatest advantages to species occupying marginal habitats, such as high mobility and multiple life histories (Thorpe 1994), may actually be selected against in isolated habitats. Consequently, without the chance for recolonization, population extinctions in fragmented stream systems may proceed in a "ratchet-like" manner, increasing the chances of basin-wide extinction (Dunham et al. 1996).

Translocation of trout into fish-less reaches above natural barriers is a common management action to increase the range of native fishes (Harig 2000). The general absence of fish from high elevation reaches above fish barriers found in this and other studies of cutthroat trout (Dunham et al. 1996, Kruse et al. 1997) indicate that this action may not assure the long-term viability of cutthroat trout populations (e.g., Harig 2000). However, our results also indicate that dispersal barriers may effectively protect WCT populations when located relatively low within stream networks. While isolation carries risks associated with low population sizes and limited physical space, it is often the only factor preventing displacement by nonnative salmonids through competition and hybridization. For example, Hanzel (1959) found that most pure cutthroat trout populations in Montana occurred above fish dispersal barriers. Young et al. (1996) reported that 20 of 27 allopatric populations of genetically pure Colorado River cutthroat trout considered indigenous, and in a drainage not recently stocked, were located above fish migration barriers. Distribution patterns we observed in the Gravelly Range illustrate the importance of natural barriers to remaining WCT populations in the Madison River drainage. Primarily because of their association with barriers occurring relatively low within stream networks, isolated populations of WCT occupied greater stream lengths and reached significantly higher abundances than non-isolated cutthroat trout populations. However, based on minimum habitat requirements suggested by other researchers (e.g., Harig 2000, Hilderbrand and Kershner 2000) the viability of most WCT populations in the Madison River drainage remains tenuous and, where possible, these

Table 2. Mean fish abundance (>75 mm total length) per linear meter of stream used for the minimum stream length estimator (MSL), and observed occupied stream lengths (including inhabited tributaries) for WCT populations sampled in this study. Bold streams meet the MSL recommended by Hilderbrand and Kershner (2000).

Stream	Mean abundance[a] fish/m	Occupied length km	MSL km
Arasta Creek	<0.10*	2.4	>25.0
Cabin Creek	**0.31**	**20.0**	**8.1**
English George Creek	0.25	9.0	10.0
Horse Creek	0.20	7.1	12.5
Hyde Creek	0.36	2.7	6.9
Papoose Creek	0.10	2.4	25.0
South Fork Indian Creek	<0.10*	2.4	>25.0
Soap Creek	0.21	3.4	11.9
Standard Creek	**0.28**	**12.1**	**8.9**
Trail Fork Bear Creek	<0.10*	<1.0	>25.0
Wall Creek	0.20	4.8	12.5

[a] From multiple pass depletion estimators except * where no estimates were made because very few fish were captured.

populations should either be expanded further downstream or replicated in larger drainages provided that potential hybridizing and competing species are first removed.

Unfortunately, in Madison River tributaries even some populations isolated by dispersal barriers were slightly introgressed, indicating that nonnative trout have been widely introduced into headwater habitats throughout the drainage. The degree of genetic introgression that can occur before the unique characteristics of WCT are no longer diagnostic for the subspecies is unknown (USDI Fish and Wildlife Service 1999). We adopted guidelines of Montana Fish, Wildlife and Parks and considered all WCT populations with 90 percent or greater purity (Montana Fish, Wildlife and Parks 1999). We feel this is an appropriate approach to analysis of fish distributions in the Madison River basin for two reasons. First, populations identified as slightly introgressed from a genetic sample may contain significant numbers of genetically pure individuals due to the nature of genetic sampling (Montana Fish, Wildlife and Parks 1999). This can result because genetic samples contain a few hybrid individuals mixed with genetically pure individuals or from testing problems related to low sample sizes (Appendix A). Consequently, slightly hybridized populations can indicate suitable habitat for WCT and may have genetic value for future conservation efforts (Montana Fish, Wildlife and Parks 1999). Secondly, the genetic status of many WCT populations in the Madison River drainage remains somewhat uncertain due to the possibility that some of these populations may contain a "deviant allele" that is a diagnostic allele characteristic of rainbow or Yellowstone cutthroat trout but that may simply be a rare WCT genetic variation (Appendix A). This situation likely exists for populations in upper English George, Cabin, Papoose and Wall creeks, and may exist for Soap Creek (Appendix A). Additional genetic sampling will be necessary for some of these populations to clarify their genetic status before population expansion or replication efforts are undertaken.

Stream Temperature

In addition to dispersal barriers, stream temperature also influenced WCT distribution in the Madison River drainage. The association of most WCT populations with fish dispersal barriers in the Madison River drainage obviously confounds our ability to make direct temperature or species interaction inferences. Overall temperature relationships would become clearer and stronger in areas where WCT and other salmonids segregated without the influence of barriers. However, in at least one stream, longitudinal temperature patterns coincided with fish distribution boundaries. Genetic data from Papoose Creek suggest that WCT segregated from nonnative salmonids without the presence of a fish dispersal barrier. Stream temperatures in Papoose Creek were significantly lower in the reach occupied by WCT than in downstream reaches occupied by nonnative salmonids. Stream temperatures in the reach of Papoose Creek occupied by WCT also were significantly lower than the coldest site occupied by nonnative salmonids in the Madison River drainage. This situation also may occur in South Fork of Indian Creek (see Fig. 2), where genetically pure WCT were observed upstream from reaches occupied by nonnative species. Unfortunately, the remote locality and lack of a priori knowledge of fish distribution and genetic status precluded temperature measurement in this stream.

Similar to our findings, Mullan et al. (1992) found that in naturally sympatric populations, rainbow trout excluded the first two or three age classes of WCT up to a point where stream temperatures decline to about 1600 annual thermal units (sum of average daily temperatures [°C]). These distribution boundaries may be attributable to temperature-mediated competitive differences between cutthroat trout and nonnative salmonids or temperature mediated growth differences. For example, Destaso and Rahel (1994) found that a 1 °C difference in Critical Thermal Maxima between brook trout (*Salvelinus fontinalis*) and cutthroat trout correlated with greater competitive ability of brook trout at warmer temperatures. Adams (1999) found that lower growth and fecundity, and greater female age-at-maturity resulting from cold stream temperatures limited upstream invasions of brook trout in some Rocky Mountain streams. In addition to brook trout, thermal tolerances of cutthroat trout are generally lower than other nonnative species such as rainbow and brown trout (Feldmuth and Erikson 1978, Eaton et al. 1995).

Although there was considerable overlap, and despite the confounding influence of dispersal barriers, all stream temperature metrics tested were significantly lower at sites occupied by WCT than at sites occupied by nonnative salmonids (Table 1). When sites occupied by rainbow trout, a potential hybridizing and competing species, were compared to sites with WCT all temperature metrics remained significantly lower for sites with cutthroat trout. The distribution of rainbow trout coincided with a 1-3 °C warmer range of stream temperatures than occupied by WCT. Magnuson et al. (1978) considered the "fundamental thermal niche" for fishes to encompass 4 °C, and Christie and Regier (1988) suggested this niche ranged from –3 to +1 °C around a species optimal growth temperature. For rainbow trout, maximum growth occurs at approximately 17.2 °C (Hokanson et al. 1977), and thus the range of maximum daily temperatures occupied by rainbow trout in the Madison River drainage (12.3-18.4 °C) corresponded closely with their fundamental thermal niche (14.2-18.2 °C). Conceivably, competitive advantages of rainbow trout at higher temperatures near their optimal growth range may account for the absence of WCT where rainbow trout were found.

Contrary to patterns for rainbow trout, no statistical differences were found between sites occupied by rainbow x cutthroat trout hybrids and those occupied by WCT at the basin level. The influence of genetic introgression of both rainbow trout and Yellowstone cutthroat trout on the thermal response of WCT has not been studied. In laboratory experiments, Ihssen (1973) found that two reciprocal first-generation hybrids of brook trout and lake trout (*Salvenlinus namaycush*) had similar times to death upon exposure to several lethal high temperatures for a series of acclimation temperatures. Second generation hybrids were intermediate to the parent species in resistance and the backcrossed offspring were intermediate between the second generation hybrids and their respective parents. This suggests that differences in thermal responses between potentially hybridizing species may quickly break down when hybrid swarms develop. If these patterns are similar for rainbow x cutthroat trout hybrids, there is a need to differentiate relatively pure from hybridized populations when investigating relationships between cutthroat trout distribution and stream temperature. Some populations with relatively high (but <90%) proportions of WCT genetic material were classified as nonnative salmonids, which may have weakened relationships between stream temperatures and fish distribution.

Basin-wide, WCT were associated with habitats where average daily stream temperatures generally remained below 12 °C and maximum daily stream temperatures remained below 16 °C. Bell (1984) reported a preferred temperature range of 9-12 °C for cutthroat trout. Dwyer and Kramer (1975) reported the greatest scope for activity in cutthroat trout occurred at 15 °C when tested at 5, 10, 15, 20, and 24 °C. Assuming that the scope for activity was a better measure of optimal temperature than temperature preference tests, Hickman and Raleigh (1982) selected 12 to 15 °C as an optimal temperature range for cutthroat trout. Average and maximum daily water temperatures at sites occupied by WCT generally corresponded with these reported ranges of preferred and optimal temperatures for cutthroat trout.

While WCT were associated with habitats where stream temperatures seldom exceeded 16 °C, this should not be construed as the upper thermal tolerance limit for this subspecies. Although warm stream temperatures approaching 25 °C appeared to limit the downstream distribution of WCT in the English George sub-drainage, temperatures in most reaches now occupied by nonnative species were well below reported critical thermal maxima of 27-28 °C for cutthroat trout (Feldmuth and Eriksen 1978, DeStaso and Rahel 1994). The patterns of fish occurrence and stream temperature we observed indicate that WCT have been displaced from warmer stream habitats and that WCT now occupy a narrower and colder range of stream temperatures than they did historically. In our study, isolated populations of WCT encountered a higher and greater range of average summer stream temperatures than in streams without dispersal barriers, indicating that without the influence of fish barriers the range of stream temperatures occupied by WCT would be substantially narrower and colder due to the influence of nonnative salmonids.

While many researchers have focused on the role that maximum stream temperatures play in regulating salmonid distribution (e.g., Dunham et al. 1999, Haas, in press), few have explicitly addressed the ecological costs for salmonids in habitats where stream temperatures remain below thermal optima. Several WCT populations sampled in this study inhabited streams where water temperatures remained below optimal temperature ranges (Hickman and Raleigh 1982) for most of the summer season (Fig. 4). Low WCT densities in Papoose Creek and Trail Fork of Bear Creek (range= 3 to <1 fish/ 100 m) may be attributed to low stream temperatures, because maximum stream temperatures remained below 10 °C throughout the summer at sites where WCT were captured in these two streams.

The two major external factors controlling fish growth are water temperature and food availability (Weatherly and Rogers 1978). Averett (1963) documented higher growth rates for WCT from lower versus higher elevation tributaries of the St. Joe River, Idaho, presumably a result of differences in stream temperatures. Body size is strongly related to fecundity in WCT (Downs 1995). Cold stream temperatures can delay cutthroat trout spawning, prolong egg incubation (Behnke 1992, USDI Fish And Wildlife

Service 1998, Harrig 2000), and reduce embryo survival (Hubert et al. 1994, Stonecypher et al. 1994). Late hatching fry risk winter starvation if they cannot grow enough to withstand metabolic deficits at low winter temperatures (Cunjak and Power 1987, Shuter and Post 1990, Harrig 2000). Consequently, WCT probably experience lower individual fitness and reproductive success in habitats where temperatures remain well below optimal ranges. The low abundances of WCT we observed at sites not physically isolated from nonnative species suggest that, while colder stream temperatures may provide a competitive or demographic boost for WCT relative to nonnative species, sub-optimal thermal regimes may also limit a population's ability to buffer environmental and demographic stochasticity in headwater habitats.

In addition to fish dispersal barriers, other local factors may affect the correspondence between fish distributions and temperature within streams, including variability of habitat quality, disease, food availability, and water quality and quantity (Dunham 1999). The potential for seasonal migrations may also add noise to data relating fish distributions directly to stream thermal characteristics (Dunham 1999). Northcote (1992) noted that the most extensive movements in resident salmonid populations were associated with spawning migrations. However, Downs (1995) reported that WCT living in headwater habitats did not appear to have extensive spawning migrations. Similarly, Shepard et al. (1998) found that while some individual WCT move relatively long distances, little movement was observed for most resident WCT inhabiting headwater streams in Montana.

A potential problem with our study is a lack of temporal concordance between fish distribution and temperature data. We matched fish sampling records with temperature records corresponding most closely in time. Because fish were sampled over a 3 year period, while most temperature data were collected in 1999, stream temperatures were not measured during the same year as the fish sampling event in some locations. For the temperature associations presented in this study to be valid, two assumptions must be met. First, fish distribution boundaries did not change during the study period. Other studies have found that distribution limits of cutthroat trout were relatively constant across a 20-year period (1977-1997) despite fluctuations in densities (Dunham et al. 1999). Similarly, brook and rainbow trout showed no net change in distribution limits over a similar time period in eastern Tennessee streams (Strange and Habera 1998). We expect this to be true in the Madison River drainage as well, especially considering the relatively short time period of our study and the strong influence of dispersal barriers on fish distribution. The second assumption is that measured stream temperatures are representative of temperatures experienced by fish during the year fish distribution data were collected. This assumption also seems reasonable because stream temperature is highly correlated with air temperature (Stefan and Preud'homme 1993) and published air temperature records for the period of our study indicate that annual and summer air temperatures from 1997-1999 corresponded closely with long term average air temperatures (NOAA 1997, 1998, 1999).

Conclusion and Recommendations

This study provided important information on the distribution and abundance of WCT in the Madison River drainage. Identifying and protecting existing populations is the first step in an effective conservation plan for WCT (Montana Fish, Wildlife and Parks 1999). Because of the inherent risks associated with the restricted distribution and small sizes of many WCT populations, simply maintaining the status quo will probably not be sufficient to ensure the long-term persistence of all populations. Due to the limited number of genetically pure populations of WCT in the Madison River drainage, we believe it would be worthwhile to replicate pure populations from Cabin and Papoose creeks. We recommend that further genetic testing be completed in English George, upper South Fork Indian, and Wall creek sub-drainages to confirm the presence of genetically pure populations in these areas. Should any of these populations prove to be genetically pure, they should be replicated, preferably somewhere within the Madison River drainage, as soon as technically feasible to conserve these unique genetic resources. When WCT populations are to be expanded, results from our study agree with others (Montana Fish, Wildlife and Parks 1999, Harrig 2000, Hilderbrand and Kershner 2000) suggesting that translocation sites be located relatively low within stream networks to ensure that habitat space and quality are sufficient to maintain the long-term viability of cutthroat trout populations. Additionally, we recommend that existing genetically pure populations of WCT be expanded downstream, where possible, to incorporate larger habitat areas. We also recommend that slightly introgressed (<10% introgression) WCT populations be managed with the same protection given to genetically pure WCT, because such populations may have genetic value and their presence indicates suitable habitat for WCT (Shepard et al. 1997, Montana Fish, Wildlife and Parks 1999).

Our study also provided important information on the thermal regimes associated with suitable habitat for WCT as well as evidence that distribution boundaries between WCT and nonnative salmonids are related to stream temperatures. However, relationships between WCT distribution and abundance and stream temperature need to be clarified through both laboratory experiments and more extensive field studies. Because thermal gradients are important determinants of species distributions and because temperature data acquisition can be costly and time consuming in remote or expansive study areas, numerous regional- or basin-scale stream temperature models have been developed recently (e.g., Keleher and Rahel 1996, Isaak and Hubert 2001, Sloat 2001). When the thermal requirements of WCT are better known, these stream temperature models can be used to prioritize WCT conservation efforts at broad scales by 1) predicting WCT occurrence in areas where their distributions are unknown, 2) identifying stream reaches where translocations of WCT have a high probability of success, and 3) predicting effects of land use and global warming on WCT distribution and abundance.

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Appendix A. Genetic testing results for sites in the Madison River drainage by date, location (legal or stream kilometer), sample size (*n*), and analysis method (E = allozyme electrophoretic and P = PINE DNA), showing species code (RB = rainbow trout; WCT = westslope cutthroat trout; and YCT = Yellowstone cutthroat trout) and proportion of sample estimated to contain alleles characteristic of each species (NA = proportions not available), and, where applicable, number of individuals that were pure WCT. Information from the Montana Resource Information System database (*http://www.nris.mt.us)* unless otherwise denoted.

STREAM Date	Location	*n*	Analysis method	Genetic Results (species code and %) Code %	Code %	Code %	Number pure WCT
ARASTA CR							
7/26/1995	07S03W36[1]	5	E	WCT 100			5
7/20/1999	08S02W06[2]	1	P	WCT NA		YCT NA	0
BUFFALO CR							
7/26/1995	07S02W31[1]	4	E	WCT 100			4
7/20/1999	08S02W05[2]	7	P	WCT 84	RB 4	YCT 12	0
7/20/1999	07S02W31[2]	7	P	WCT 84	RB 4	YCT 12	0
CABIN CR							
8/31/1997	11S04E05	7	E	WCT 100			7
11/15/1998	11S03E15	8	E	WCT 0	RB 71	YCT 29	0
4/19/1999	11S03E14	10	P	WCT 93	RB 7	YCT 0	0
7/26/1999	Km 3.2-9.3	27	P	WCT 96	RB 4	YCT 0	0
7/27/1999	11S04E14[3]	6	P	WCT>90	RB<10		5
CABIN CR, M FK							
6/01/1993	11S04E11	10	E	WCT 100			10
7/27/1999	Km 0-8.0[3]	58	P	WCT 98	RB 2	YCT 0	0
CORRAL CR							
7/8/1998	Km 9.7[4]	21	P	WCT 86	RB 8	YCT 6	0
ENGLISH GEORGE CR							
8/1/1992	09S01W36	15	E	WCT 95	RB 5	YCT 0	0
6/8/1999	10S01W02[5]	10	P	WCT>90	RB<10		*NA*
HORSE CR							
8/10/1995	10S02W19[1]	8	E	WCT 100	RB 0	YCT 0	8
7/28/1998	Km 7.2-11.3[6]	70	P	WCT 88	RB 3	YCT 9	0
7/28/1998	Km 12.1-13.7[6]	29	P	WCT 98	RB 0	YCT 2	*NA*
HYDE CR							
7/21/1995	09S01W34[2]	3	E	WCT 96	RB 4	YCT 0	0
7/13/1999	09S01W33[3]	16	P	WCT 96	RB 0	YCT 4	0
MIDDLE FORK BEAR CR							
7/27/1994	07S02E06[7]	2	E	WCT 87	RB 13	YCT 0	0
NORTH FORK BEAR CR							
7/26/1994	07S01E36[7]	4	E	WCT 70	RB 25	YCT 5	0
PAPOOSE CR							
7/26/1994	11S02E06[7]	4	E	WCT 100	RB 0	YCT 0	4
7/27/1999	Km 0-5.6[8]	24	P	WCT NA	RB NA	YCT NA	6

Appendix A. (continued)

STREAM Date	Location	*n*	Analysis method	Genetic Results (species code and %) Code %	Code %	Code %	Number pure WCT
QUAKING ASPEN CR							
6/30/1998	Km 1.6	16	P	WCT 77	RB 23	YCT 0	0
SOAP CR							
9/19/1991	11S01E29	12	E	WCT 99	RB 0	YCT 1	0
9/01/1992	11S01E29	16	E	WCT 99	RB 0	YCT 1	0
SOUTH FORK ENGLISH GEORGE CR							
6/8/1999	10S01W02[9]	9	P	WCT NA	RB NA	YCT NA	*NA*
SOUTH FORK INDIAN CREEK							
8/05/1998	Km 1.6-4.0[10]	22	P	WCT 79	RB 15	YCT 6	0
8/05/1998	Km 4.0-5.6[10]	12	P	WCT >90	RB NA	YCT NA	*NA*
STANDARD CR							
8/11/1997	11S01E05[11]	13	E	WCT NA	RB NA	YCT NA	0
TEPEE CR							
8/01/1995	10S02W 13[1]	5	E	WCT 100	RB 0	YCT 0	5
7/28/ 1998	Km 1.6	13	P	WCT 98	RB 0	YCT 2	0
WALL CR							
7/13/1999	Km 5.6	7	P	WCT 97	RB 0	YCT 3	0
WIGWAM CR							
7/20/1999	08S02W 07[2]	7	P	WCT 82	RB 1	YCT 17	0

[1] Information from letter to Jim Brammer, Montana Fish Wildife and Parks (MFWP), from Robb Leary, University of Montana Wild Trout and Salmon Genetics Laboratory (WTSL) dated May 6, 1997.

[2] Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000. Samples from locations in Buffalo Creek combined. An individual trout collected from Arasta Creek possessed PINE markers characteristic of both westslope and Yellowstone cutthroat trout but proportions were not available.

[3] Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated August 21, 2000. A single allele characteristic of rainbow trout was present in one fish from Cabin Creek at T 11, R S04E, SEC 14, indicating either slight genetic introgression or a pure westslope cutthroat trout with a single deviant allele similar to rainbow trout.

[4] Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated November 8, 1999.

[5] In English George Creek a single allele characteristic of rainbow trout was present at low frequencies. This could indicate a small amount of hybridization or it could simply be a rare westslope cutthroat trout genetic variation. Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000.

[6] Within the Horse Creek drainage (Horse and Tepee creeks) all fish were hybridized between westslope cutthroat, Yellowstone cutthroat, and rainbow trout, however, the population above a waterfall near stream mile 7.5 did not contain any rainbow trout alleles, had what may have been a few pure westslope cutthroat trout individuals, and had a higher proportion of westslope cutthroat trout alleles than the population below the falls. Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated November 8, 1999.

[7] Information from letter to Jim Brammer, MFWP, from Robb Leary, WTSL, dated May 23, 1995.

[8] Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000. Proportions not available. Some fish that were pure WCT (6 of 9) were sampled at 4.0, 4.8, and 5.6 km. All fish below 4.0 km were either rainbow (3 of 15) or hybrids. The three hybrids above 4.0 km contained a single allele characteristic of RB.

[9] In the South Fork English George Creek, a single allele characteristic of Yellowstone cutthroat trout was present in one individual. It may be a pure westslope cutthroat trout population with a single deviant allele that is similar to Yellowstone cutthroat trout. Additional sampling is necessary. Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000.

[10] Fish from the South Fork Indian Creek were all classified as hybrids between westslope cutthroat, rainbow, and Yellowstone cutthroat trout. However, fish from stream kilometer 4.0 to 5.6 contained over 90% westslope cutthroat trout alleles, while fish from lower in the drainage contained much lower westslope cutthroat trout allele frequencies. Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated November 8, 1999.

[11] Information from letter to Brad Shepard, MFWP, from Naohisa Kanda and Robb Leary, WTSL, dated November 2, 1998. While westslope cutthroat trout genes were dominant, some Yellowstone cutthroat trout introgression was documented. A freezer malfunction made it impossible to determine the extent of introgression with either Yellowstone cutthroat or rainbow trout.

Development Of An International Conservation Strategy For Burbot In Idaho And British Columbia

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Abstract

Burbot (*Lota lota*) once provided popular sport, commercial, and subsistence fisheries in the lower Kootenai drainage of Idaho and British Columbia. However, the respective burbot fisheries collapsed in the late 1970s and closure of the fisheries provided no improvement to the burbot populations. Research indicated one of the primary problems affecting burbot is high peaking flows released from Libby Dam during winter spawning migration. Other physical and biotic changes to the ecosystem also may have played a role in the decline of burbot. The burbot in Idaho and Kootenay Lake are currently near demographic extinction. Agencies responsible for burbot management believe efforts should be made to recover this stock, and fisheries and river managers feel development of a conservation strategy could be useful to identify and prioritize actions necessary to recover burbot. The most important aspect of the conservation strategy, designed to be refined or amended as new information becomes available, may be identification of factors limiting the population. This paper provides the logic behind most categories within the burbot conservation strategy and can be used as an example in preparation of strategic documents for other species at risk. If management agencies and water use managers agree in writing to follow the prescribed measures of a conservation strategy then the strategy becomes a conservation agreement.

Key words: burbot, conservation strategy, Kootenai River, Kootenay Lake, *Lota lota*

Introduction

Most native sport fishes of the Kootenai River system are at risk or threatened with extinction, including the Kootenai River white sturgeon (*Acipenser transmontanus*) (59 Federal Register 45989 1994), bull trout (*Salvelinus confluentus*) (63 Federal Register 31647 1998), interior redband rainbow trout (*Oncorhynchus mykiss gairdneri*) (Williams et al. 1989), westslope cutthroat (*O. clarki lewisi*), kokanee (*O. nerka*) (Partridge 1983), and burbot (*Lota lota*) (Partridge 1983, Paragamian et al. 2000). Recovery plans, or conservation strategies, have been prepared for the two listed species in the Kootenai River: white sturgeon and bull trout (State of Idaho 1996, USDI Fish and Wildlife Service 1999). Recently, i.e., 2 Feb 2000, the Idaho Conservation League and American Wildlands petitioned the

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USDI Fish and Wildlife Service (USFWS) to consider an emergency endangered listing for burbot in the Kootenai River, Idaho.

Burbot fisheries in the Kootenai River of Idaho and British Columbia (BC) and Kootenay (Canadian spelling) Lake, BC, once provided important fishing and subsistence opportunities, but are now in a state of collapse (Paragamian et al. 2000). Because of their native status and historical fisheries importance, agencies responsible for management of burbot and their habitat formed the Kootenai River Burbot Recovery Committee to formulate a conservation strategy. The goal of the committee was to develop a cooperative approach to prevent any further decline and identify actions needed to rehabilitate the burbot population.

The committee subsequently developed an international conservation strategy for burbot in the Kootenai River of Idaho and BC and Kootenay Lake, BC, to facilitate conservation activities for burbot. This conservation strategy is also applicable to the Montana reach of the Kootenai River from the Idaho border upstream to Kootenai Falls, Montana; however, information regarding burbot is limited from this reach of river. Thus, this document does not fully consider burbot in the Montana reach. This document, patterned after conservation strategies prepared for Bonneville cutthroat trout (*O. clarki utah*) in Idaho and Utah, and bull trout in Lake Pend Oreille, Idaho, matured through the efforts of representatives from several state, provincial, federal, and tribal agencies. We describe here a series of elements from the original International Conservation Strategy for Burbot in the Kootenai River, Idaho, and British Columbia to provide the reader, through example, a reference or a guide toward developing conservation strategies for other species at risk.

A conservation strategy is one approach to identify actions aimed at reducing threats to a population once it is determined at risk of extinction. If management agencies agree in writing to follow the prescribed measures of a conservation strategy then the strategy can become a conservation agreement—other definitions of a conservation agreement also exist. A conservation agreement is a positive step towards avoiding federal intervention such as listing under the Endangered Species Act (ESA). Under Section 4 of the ESA it may be used in lieu of a recovery plan. Other benefits may include less regulatory imposition and bureaucracy.

In this particular case, a conservation strategy was believed necessary for several reasons: 1) the burbot population in the Idaho and British Columbia portion of the Kootenai River is at risk of demographic extinction and considered genetically different than burbot in the upper river above Kootenai Falls, Montana (Paragamian et al. 1999); 2) Libby Dam, built on the Montana reach of the Kootenai River in 1972, has a complex set of operating protocols, e.g., flood control and hydropower, that did not originally consider biological effects and may have significantly influenced the decline of burbot in the river downstream (Paragamian et al. 2000); 3) federal agencies involved in river management, e.g., U. S. Army Corps of Engineers (USACE) and Bonneville Power Administration (BPA), had no compelling mandate to move forward to rehabilitate the habitat in the Kootenai River to aid burbot conservation; and 4) there is the prospect of a listing action under the ESA and some state, federal, provincial, and tribal managers believed it important to have a conservation strategy in place if and when the opportunity to implement a conservation agreement arose. If the population were listed as endangered, a recovery team would be formed to develop a recovery plan. An existing conservation strategy for burbot would be a useful foundation for a recovery plan and hasten implementation of recovery efforts. Such a plan would give state, tribal, and provincial agencies more responsibility for implementation of conservation measures and a greater role in the federal listing and recovery processes.

The Conservation Strategy Committee

Conservation committees can be organized by and comprised of anyone with a dedicated interest in preparing a cooperative management document to restore or rehabilitate a population at risk of extinction or to reduce a threat to the population's integrity. The senior author organized and chaired the Kootenai River Burbot Committee comprised of state, federal, provincial, and tribal agency biologists and managers with technical expertise to contribute to specific areas of need in the document or represent agencies and organizations with a vested interest in the persistence of burbot. The chairman scheduled and conducted meetings at approximately six-month intervals. First the structure of the conservation strategy was studied, and then assignments were made, which emphasized areas of expertise, to prepare specific segments of the document.

Local support may be one of the most important attributes to success of a conservation strategy. Addition of members of local community government to a conservation strategy committee may benefit the process by providing a historical and socioeconomic perspective as well as political or local public support. Although local governments knew of our efforts to prepare a conservation strategy for burbot we brought them into the process only after it was complete. Our experience later suggested that when local individuals directly impacted by management changes—water management in our case—are included at some stage of the process, preferably at conception, they likely will take direct ownership in it, provide support, and may eventually identify with its success. In addition, concern regarding listing under the ESA, and associated loss of local control of the process (either real or perceived) also provides local governments an incentive for involvement.

Certainly with the myriad of agencies and personalities active on a conservation strategy committee, disagreements regarding document wording, obligations, jurisdictional or management authorities and responsibilities, or the development of appropriate management actions or measures are inevitable. For example, during preparation of the Kootenai River Burbot Conservation Strategy, two federal agencies were hesitant to fully engage until assured that the conservation strategy had no legally binding authority. It is the responsibility of the chairman to guide resolution of disagreements and clarify misunderstandings. Generally, producing a legally binding conservation agreement based on the conservation strategy is more difficult because managment agencies must disclose their true position and make financial, logistical, or operational concessions to facilitate recovery/ rehabilitation efforts.

The Conservation Strategy

Life History

This section lays out a framework of the basic life history characteristics of the species of concern. Important components include critical habitat needs of each life history stage and where the candidate species fits into the trophic structure of the fish community. Of particular importance are factors that make the species unique or vulnerable to habitat alterations or other human-caused influences. In the case of burbot in the Kootenai River, important life history characteristics include extensive winter spawning migrations, highly synchronized migration and annual maturation, low swimming endurance, and potential impacts of changes in water temperature. The objective is to clearly tie these life history characteristics and any changes in population status, based on scientific evidence, to environmental disturbance or natural change that directly impacts the population.

Burbot are the only freshwater member of the family Gadidae, the cod family, reaching a weight of about 4.5 kg in Idaho and BC. Burbot are winter spawners, spawning at temperatures usually below 4 °C (McPhail and Paragamian 2000) and are highly synchronized in both gonadal development on an annual basis and arrival to a spawning site (Arndt and Hutchinson 2000, Evenson 2000). Burbot spawn in large groups called "spawning balls" and a female burbot may release over one million eggs (Becker 1983). The eggs are semi-buoyant and drift downstream to a lake to hatch and rear. Very young burbot are pelagic and feed on zooplankton. As they grow in length (≥15 mm) they move to the shoreline and feed on insects and small fish. After the first year of life they move into deeper water, feeding on fish and Mysid shrimp (*Mysis relecta*). Male burbot mature at about age-3 and age-4 while females mature at about age-4 and age-5. Burbot have low stamina and swimming endurance (Jones et al. 1974). Velocities >25 cm/s affected sustained swimming endurance when subjected for >10 minutes (Jones et al. 1974). Thus, it is reasonable to believe increased flow at critical periods could affect spawning success.

Status and Distribution

This section should provide a historic background of the abundance, distribution, and recreational importance of the species, as well as information of cultural and social importance. Temporal changes, especially declines in regards to any of these aspects, are important to frame the conservation need. In the case of burbot in the West Arm of Kootenay Lake, a creel survey over 20 years provided chronological evidence of the decline in the burbot fishery. Fishery surveys also can provide valuable information on the timing of significant population change. A discussion of distribution of the species of interest may support categorization of a population as of special concern or rare status. Distributional information also helps define geographic, management, and jurisdictional boundaries. Anecdotal information can be used with caution to describe population trends.

Idaho.—Although burbot are circumpolar in distribution, including the northern tier of states, Alaska, and most of Canada (McPhail and Paragamian 2000), they are endemic only to the Kootenai River in Idaho (Fig. 1; Simpson and Wallace 1982). The earliest records of burbot sampling in the Kootenai River, Idaho were taken from the Idaho Department of Fish and Game (IDFG) Panhandle Region archives during the 1957-1958 winter. The combined sport and commercial fisheries in Idaho during the 1950s and 1960s are thought to have yielded >2000 kg/year. Soon after completion of Libby Dam, in 1972, a substantial reduction in the abundance of burbot was noted. After dam closure, Partridge (1983) captured a total of 108 burbot from 1979 through 1982; however, a follow-up study of burbot captured only 17 burbot in 1993 (catch/unit effort [CPUE] of one burbot/33 net days) and eight in 1994 (CPUE of one burbot/111 net days). Numerous age groups of fish were represented in the net catch, indicating some burbot recruitment. Recruitment was apparently coming from the Goat River in BC, the only known spawning location (Paragamian 1995; Fig. 1). We found no evidence of reproduction in Idaho.

The historical burbot fishery in Idaho was primarily a winter fishery although some fishing occurred in spring. Following construction of Libby Dam, warmer winter water releases from Lake Koocanusa, formed by the dam, eliminated ice cover and the associated ice fishing on the Kootenai River (Partridge 1983). Burbot regulations in Idaho were liberal, i.e., no closed season or bag limit, until 1984 when the state adopted a two-fish/day limit. A ban on all burbot harvest followed in 1992.

Kootenay River and Lake, British Columbia.— The burbot population in Kootenay Lake and the Kootenay River in BC are red-listed—identified as threatened or at risk of extinction—as an imperiled population in BC. Historically, the primary fishery for burbot was in the West Arm of Kootenay Lake near Balfour, BC, and was primarily a late spring-early summer fishery.

In the West Arm of Kootenay Lake

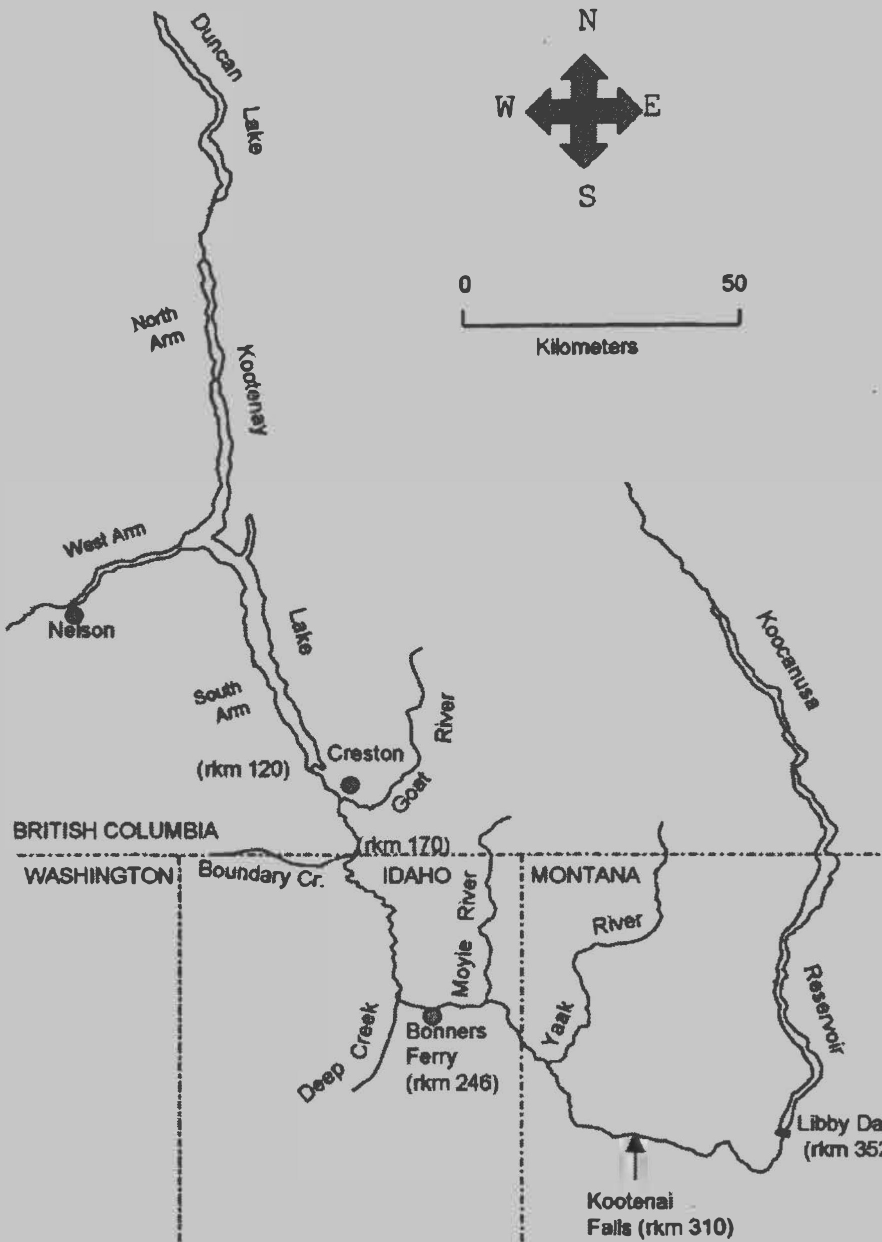

Figure 1. Kootenai River and locations of Bonners Ferry, Libby Dam, Lake Koocanusa, Kootenay Lake, and other important locations , including river kolometer (rkm).

over 26,000 burbot were caught in 1969 and about 20,000 in 1971. The angling catch rate of burbot averaged about one fish/hour during the 1969-1971 period. Until the mid-1960s, the daily harvest limit was 15 burbot/day but was lowered to 12/day in 1967 (Sinclair and Crowley 1969). Harvest of burbot declined substantially in years following 1971, and angling overharvest was a concern. As a result, the daily limit was reduced to 10 burbot/day in 1975 (Andrusak 1976) and to 5 fish/day in 1976 although no corresponding improvement in the burbot population was observed. Harvest of burbot continued to decline through the 1970s with angling catch success remaining at about 0.7 fish/hour. As of 1987, no fish were recorded in the West Arm fishery and in 1997 the lake was closed to burbot fishing. We know little of the historical fishery in the Kootenay River, BC.

Kootenai Tribe and burbot fishing.— The Kootenai Tribe historically relied heavily upon burbot to provide a dietary staple and they were very adept at using weirs and traps to capture burbot. The following information was provided by the late Abe Abraham, a well respected Kootenai Tribal Elder: *" The ling moved into the tributaries to the Kootenai River from Kootenay Lake all the way to the mouth of the Moyie River. We fished for ling through the ice in February using lines and large hooks baited with squawfish and peamouth chub. Ling were extremely abundant and were a main staple for the Tribe in the late winter/early spring months. They were the most important food source for the Tribe at that time of year."* Additionally, the Tribe recognizes the connection of all resources in the web of life, and that the preservation of all native fish is important to the ecosystem.

Principle Habitat Concern

An important segment of the conservation strategy is to explain what is known of the major threats to the population, which can and often does include habitat changes. For burbot in the Kootenai River, it is important to understand the operational procedures for Libby Dam because hydro operations are thought to have significantly affected habitat in the river and contributed to decline in the burbot population. Understanding how the habitat has changed or, in this example, the agent of change—Libby Dam—can provide insight into the effect on the ecosystem and native species, as well as how effects may be modified or mitigated to lessen their impact. For example, an alternative management plan for the operation of Libby Dam called VarQ (U.S. Army Corps of Engineers 1999) attempts to include flood control operational guidelines that are believed, when compared to past guidelines, more considerate of sturgeon and salmon needs, and possibly burbot as well. A conservation strategy for one species, i.e., burbot, should adopt habitat and flow recommendations that are compatible with any exisitng recovery plan of conservation strategies for other species. An additional consideration when considering habitat or operational modifications in a conservation strategy is compatibility with other Recovery Plans or Strategies of other species at risk, e.g., white sturgeon, that are already in place. Ideally, conservation strategies should complement each other.

Libby Dam on the Kootenai River in Montana is authorized primarily for flood control and power production, and is operated by the USACE. Operating guidelines for endangered fish populations are specified in the 1995 and 2000 Biological Opinion for Kootenai River white sturgeon (Dwyer 1995, USDI Fish and Wildlife Service 2000), the 1995 Biological Opinion for Snake River salmon (National Marine Fisheries Service 1995), the 1998 Biological Opinion for Columbia/ Snake River steelhead (National Marine Fisheries Service 1998), a 2000 Biological Opinion for the listed bull trout (USDI Fish and Wildlife Service 2000), and annual guidelines from the USFWS (National Marine Fisheries Service 1998). From April until July each year, USACE strives to fulfill the sometimes-conflicting objectives of flood control and flow augmentation for endangered Kootenai River white sturgeon and endangered salmon populations downstream. A federal project such as Libby Dam is often forced to confront the sociopolitical conflicts of species recovery vs. human needs or expectations, and agency mandates. For example, increased flows for sturgeon or salmon are thought by some to be a threat to agricultural grounds and the local economy. At Libby Dam, the flood control and power operation takes precedence over sturgeon recovery and salmon needs, which is based on providing 200-year flood protection.

More recently negative consequences for burbot have become apparent (Paragamian 2000) during the critical winter spawning months due to the operation of Libby Dam. During winter the dam is operated to draw down Lake Koocanusa in preparation for spring runoff, as well as to maximize winter power revenue; thus, winter river velocities are substantially higher than pre-dam conditions at a time when burbot are migrating and spawning in the Kootenai River. As previously noted, burbot are ill equipped to withstand the higher velocities resulting from dam operation.

Problems Significantly and Negatively Affecting Candidate Species

Another important aspect of a conservation strategy is identifying factors most limiting to a population or species of concern. Development of this section should include an exhaustive literature review and discussion of research results that identify limiting factors directly linked as a threat to the species or a specific portion of its range, e.g., population. The narrative then synthesizes and explains how the factors may have impacted the species or population of concern, including shortcomings or gaps in available information. Including published peer-reviewed studies adds credibility to the strategy. Presenting categories in an orderly manner also provides a measure of logic. This section can be refined or amended as new information becomes available, but should rely on the best available information at the time of preparation of the final conservation strategy. Understanding the limitations is critical for developing recovery strategies and their expected outcomes.

Dam operation and reservoir management.— Operation of Libby Dam for hydropower, which included power peaking until 2000 when it was discontinued, and flood control changed the river's natural hydrograph from low flows in winter and high flows in summer to post-dam flows that are comparatively lower in summer and higher in winter. The release of water impounded in Lake Koocanusa also has elevated winter water temperatures and reduced summer temperatures (Partridge 1983, Paragamian et al. 2000).

Prior to construction of Libby Dam winter, the time when burbot migrate and spawn, was the most environmentally stable period of the year. Burbot travel over 125 km to spawn (Bresser et al. 1988). Burbot in the Kootenai River are thought to have traveled ≤120 km from Kootenay Lake to spawn in tributaries in Idaho (Paragamian 1995). Elevated winter flows and water velocities from Libby Dam have disrupted burbot spawning migration, in terms of the timing of activity and upstream passage (Paragamian 2000). The specific effect of this disruption to burbot spawning is unknown, but it may have diminished spawning fitness, changed spawn timing, reduced stamina, or even impacted reproductive fitness. Any one of these impacts could collapse the fishery by reducing spawning success and recruitment (Paragamian 2000, Paragamian et al. 2000). Fluctuating flows from Libby Dam appear to provide confusing migration cues and disrupt upstream spawning migrations by burbot (Paragamian 2000). Many burbot with sonic transmitters demonstrated interrupted sequences of upstream movement, no movement, and/or fall back during the winter of 1994-1995 and 1997-1998 that was presumably related to documented flow changes. Daily differences in flow from Libby Dam during the winter can range up to about 650 m^3/s.

Nutrient losses.— As with many reservoirs, Lake Koocanusa acts as a nutrient sink and has reduced productivity of the river downstream (Snyder and Minshall 1996, Woods 1982) and Kootenay Lake (Northcote 1973). Unpublished data indicates the loss of primary productivity due to nutrient loss in the river might have played a role in the collapse of burbot (Paragamian 1994, Aherns and Korman 2002), possibly by limiting resources available to young burbot.

River temperature.— How warmer winter temperatures in the Kootenai River may affect burbot spawning migration and spawning is unknown. During the winters of 1994-1995 Paragamian (2000), using sonic tags, found burbot were attracted to the colder water of the Goat River from the Kootenai River when temperatures were 0 °C and 4 °C, respectively. Soon after this observation burbot ascended the Goat River to spawn. However, several days later when the temperature of the two rivers was similar, three additional burbot in the Kootenai River bypassed the Goat River on a suspected spawning journey into Idaho. We have no evidence to suggest these burbot spawned. How warmer temperatures affect burbot reproductive success since they are temperature sensitive (Becker 1983) requires further research.

Conservation Goal

The conservation goal section of the conservation strategy provides a broad overview of contingencies that must be accomplished for the successful recovery/ rehabilitation of a candidate population and recognition of when the goal is accomplished. The conservation goal may provide a brief statement of why the goal is achievable, provided the recommended measures are implemented. It also reaffirms a commitment to what level of recovery/ rehabilitation will be achieved, e.g., a self sustaining population.

The goal of the burbot conservation strategy is to maintain and restore multiple life history strategies, and maintain the genetic diversity necessary to sustain a viable burbot population. Restoration of the burbot population will be considered complete or successful when monitoring and evaluation of the population indicates a healthy age-class structure and a sufficient fish density to support harvest for subsistence and sport fishing. Sustained burbot populations have environmentally stable spawning and rearing habitats, juvenile habitat, and adult habitat (McPhail and Paragamian 2000). Restoration or rehabilitation of the burbot population in the Kootenai system is contingent on significant habitat changes; for example, lower winter flows during burbot spawning migration and spawning, improved food productivity, and perhaps lower winter water temperatures.

Conservation Objectives

Conservation objectives should be established from an analysis of the factors limiting the viability of the species. An objective includes the tasks or measures necessary to reduce or eliminate one or more limiting factors. Simply put, *"What is necessary to recover/rehabilitate the candidate species?"*

For burbot recovery efforts to be successful, i.e., goal attained, the physical and biological environment required by all the species' life stages must be restored or rehabilitated and maintained. The objectives necessary to do this must be achievable, repeatable, and measurable, e.g., evidence of reproduction, increasing population numbers, or movement to spawning tributaries during winter. The burbot population must respond to the actions and measures of an objective. These objectives can be reorganized, changed, or become more specific when framing a conservation agreement. This section must provide the specific measures that are necessary to recover or rehabilitate the target species or population based on the information available when the conservation strategy is finalized. As such, the following include objectives for the burbot conservation strategy:

Ecosystem recovery.—To recover the burbot population, a more normal riverine ecosystem with less influence from Libby Dam must be restored and the loss of primary productivity must be addressed. An approach such as the adaptive ecosystem assessment (AEA) methodology, which has shown usefulness in previous studies (Walters et al. 1996, Ashley et al. 1996), could be used to model ecosystem response to manipulation of environmental variables known to be important to burbot. Such methodology could provide some insight regarding the response of the burbot population to large-scale ecosystem rehabilitation actions, such as addition of nutrients to the system to improve productivity, before costly and controversial actions are actually deployed.

Flow manipulation.—Current flood rule curves used by the USACE should be modified and winter (Jan and Feb) flow levels in the Kootenai River should be returned to pre-dam levels (~170 m^3/s), in order to facilitate the burbot spawning migration. This would provide burbot a migration corridor more similar to natural conditions and allow uninterrupted migration to tributaries in Idaho. Reduced winter flows would also equate to cooler more natural temperatures. Because the population is transboundary, these recommended changes are expected to benefit burbot in the Kootenay River and Kootenay Lake in British Columbia. VarQ, a proposed USACE hydrosystem flood control operation for the Columbia River Basin (U.S. Army Corps of Engineers 1999) should be adopted and, in combination with Kootenai Integrated Rule Curve (KIRC) (USDI Fish and Wildlife Service 1999), could result in a configuration of flows beneficial to burbot recovery (USDI Fish and Wildlife Service 1999) below Libby Dam.

Donor stocks.— Because burbot stocks in the Kootenai River have been depleted to the point of near demographic extinction, the life history attributes and population genetics of other burbot stocks in the drainage, e.g., Duncan and Moyie lakes, should be documented as potential refounders. If the Kootenai stock reaches a point where recovery is unlikely even with habitat and flow management, and extinction risk is high, the introduction of burbot of similar genetic makeup and life history may be necessary to enhance or refound burbot in the system to aid in recovery.

Confined brood stocks.— Experimental enclosures containing gravid burbot should be used to determine their affect on enhancing natural reproduction. Burbot reproduce in large schools and spawners may have a strong affinity to other spawners. Restoration of burbot may be enhanced by maintaining gravid burbot in an enclosure, to act as attractants to other potential spawners.

Burbot culture.— Culture of burbot is a new endeavor and should be considered experimental at this time. If culture is determined to be feasible for burbot, this tool should be used to prevent extinction and preserve the remaining gene pool while other measures are implemented to restore habitat conditions necessary for successful recruitment of stocked individuals.

Additional turbines at Libby Dam.— The installation of additional turbines at Libby Dam should be considered. Additional turbines would allow for more flexibility in water management, such as spring flood control, because higher flows could be passed without spilling (and associated dissolved gas problems) when the reservoir is full. Thus, an additional turbine that allowed greater flood control capabilities would reduce the concern for higher spring reservoir elevations, allowing a reduction in winter flow.

Spring management of Kootenay Lake.— The management of Kootenay Lake should be studied for potential impacts on burbot. This poorly understood aspect of the Kootenay River system could have an effect on burbot. Since construction of Libby Dam, a dam downstream of the outlet artificially controls the elevation of Kootenay Lake. The International Joint Commission Board of Control, a joint commission of the U.S. and Canadian governments, governs water elevation in Kootenay Lake. The possible impact of lowering Kootenay Lake each spring potentially threatens rearing of larval burbot in tributaries. Elimination of the spring draw-down of Kootenay Lake, drafting the lake at a much slower rate, and setting the target elevation date for Kootenay Lake back by at least one month could benefit burbot and should be examined.

Assessment techniques.— The stock status of the burbot population must be determined annually to evaluate changes due to implementation of recovery objectives. There is also a need to continue improving adult and juvenile stock assessment methods to better monitor recovery efforts and progress. However, as the burbot population decreases in number, it becomes increasingly difficult to monitor the population. In addition, any unnecessary handling may pose a threat to the survival of individual fish.

Monitoring, Compliance, and Review of Conservation Measures

This final section of the conservation strategy lays out the general responsibilities for the participating and managing agencies and is based on the assumption that a conservation agreement will be adopted, coordination will be necessary, and actions will be implemented. It should include an implementation schedule that provides a calendar of actions such as remedial measures, monitoring, and evaluation needed to fulfill the defined objectives. Monitoring and evaluation include following trends in the target population, both before and after restoration actions are initiated, and providing progress reports to the Conservation Committee regarding implementation of the overall plan and its success in recovery or rehabilitation of the species or population. Annual reports are needed to document activities, add to the developing database, and provide insight or guidance to changes that may be necessary to the conservation agreement. Most importantly, this section must explicitly define individuals or agencies that are responsible to complete, i.e., fund, identified tasks in a timely fashion, either independently or as the lead on a cooperative effort among agencies.

Conclusion

Concurrent with preparation of this manuscript, a local resource advisory council comprised of representatives from local government and other stakeholders in Boundary County, Idaho, federal, state, and Tribal agencies formed to provide a forum for natural resource issues. The Kootenai Valley Resource Initiative has been meeting since January of 2002. A subcommittee of this group was formed in April of 2002 and the first item on their agenda was the crafting of a Burbot Conservation Agreement with the USDI Fish and Wildlife Service, USACE, and BPA. The conservation agreement is based on the adoption of the International Kootenai River Burbot Conservation Strategy. A water management model (based on VarQ) was developed to demonstrate how storage and flow could be manipulated to ensure maintenance of flood control measures for the Kootenai River, but also provide water for white sturgeon spawning and rearing, salmon flows, and suitable winter low flows for burbot migration, spawning, and rearing. The proposed water management scheme would take effect during years of low and normal runoff thereby allowing other operational strategies in high flow years to alleviate flood concern. Studies also are underway to restore nutrients to the Kootenai River and improve food production.

Development of a conservation strategy can be the first step in bringing about a positive working environment and foster the establishment of a conservation agreement. Our example of a conservation strategy provided the reader with a basic plan. Each conservation strategy can differ and there is opportunity to craft each strategy in accordance with the special needs of the species at risk.

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A Native Westslope Cutthroat Trout Population Responds Positively After Brook Trout Removal And Habitat Restoration

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Abstract

The distribution and abundance of westslope cutthroat trout (*Oncorhynchus clarki lewisi*; WCT) have dramatically declined across much of their historical range, but particularly within the upper Missouri River basin of Montana. A genetically pure remnant WCT population inhabits White's Creek in the Missouri River basin, Montana; however, this population was extremely low during the early 1990s (~ 80 fish ≥75 mm long) due primarily to interaction with sympatric, nonnative brook trout (*Salvelinus fontinalis*) and habitat alterations caused by past dredge and placer mining. From 1993 to 2000 brook trout were removed by repeated electrofishing. In 1995 the mining-impacted portion of the stream was restored and a fish migration barrier was constructed. Brook trout were successfully removed from White's Creek above the constructed barrier after eight years of intensive electrofishing effort. The population of WCT increased dramatically, at least seven-fold, following removal of brook trout, with the most pronounced response seen for age-0 WCT. The portion of White's Creek where stream habitat was restored supported similar abundances of WCT (catches of ~30 WCT ≥75 mm long/100 m of stream) as found in natural stream reaches above and below this restored portion. Following brook trout removal, standing crops of WCT in allopatry were similar to combined standing crops of brook trout and WCT in sympatry prior to initiation of brook trout removal. Electrofishing appeared to be an effective tool for removal of brook trout in this small stream with relatively uncomplex habitat; however, we caution that it may not be effective in larger systems.

Keywords: brook trout, competition, electrofishing, fish barrier, habitat restoration, native trout conservation; nonnative fish removal, *Oncorhynchus clarki lewisi*, westslope cutthroat trout, *Salvelinus fontinalis*

Introduction

The distribution and abundance of westslope cutthroat trout (*Oncorhynchus clarki lewisi*; WCT) have dramatically declined across much of their historical range, but particularly within the upper Missouri River basin of Montana (Liknes and Graham 1988, Behnke 1992, McIntyre and Rieman 1995, Van Eimeren 1996, Shepard et al. 1997). Factors associated with this decline include introductions of nonnative fishes, habitat changes, and over-exploitation (Hanzel 1959, Liknes and Graham 1988, Behnke 1992, McIntyre and Rieman 1995). Genetic introgression with introduced rainbow (*O. mykiss*) and Yellowstone cutthroat (*O. c. bouveri*) trout also represents a serious threat to WCT throughout their range (Allendorf and Leary 1988). Leary et al. (1987) suggested that the subspecies WCT should be accorded the same attention given to taxonomically recognized species due to their high amount of genetic divergence. Due to a relatively high amount of genetic variability observed among populations of WCT, Allendorf and Leary (1988) recommended conservation of many populations throughout its historical range as necessary to conserve genetic diversity presently contained within this subspecies.

Shepard et al. (1997) estimated that genetically pure populations of WCT within Montana's upper Missouri basin currently occupy <5 percent of their historical range and indicated that many remaining extant populations in the Missouri basin had relatively low probabilities of persisting for the next century unless conservation measures were implemented. Montana has a long history of WCT conservation and formalized a collaborative statewide conservation agreement with federal land management agencies and several private organizations (Montana Fish, Wildlife and Parks 1999). A primary objective in this conservation agreement is the protection and expansion of existing populations.

Many habitats historically occupied by WCT now contain populations of nonnative trout and in many cases these nonnative trout have totally replaced WCT (MacPhee 1966, Griffith 1972, Behnke 1979 and 1992, Liknes and Graham 1988, McIntyre and Rieman 1995). This type of replacement has also been suggested for other cutthroat trout subspecies (Behnke 1979 and several papers in Gresswell 1988). Griffith (1988) reviewed the literature on competition between cutthroat trout and other salmonids and concluded that interactions between native rainbow trout and WCT probably resulted in either spatial or niche segregation between the two species.

In the upper Missouri basin a large proportion of historical WCT habitats are now occupied by nonative brook trout (*Salvelinus fontinalis*) introduced into the basin during the early 1900s (Shepard et al. 1998). Griffith (1970, 1972, 1974) documented dietary overlap between brook trout and WCT and suggested that brook trout could replace WCT, but this replacement likely occurs only after degradation of habitat has reduced or eliminated WCT. Thomas (1996) observed that young brook trout inhibited foraging efficiency of juvenile Colorado River cutthroat trout (*O. c. pleuriticus*) in a controlled laboratory setting. She suggested that this inhibition might be the mechanism responsible for decreased growth rates she documented for cutthroat trout in the wild. Underwater microhabitat observations on positions occupied by brook trout and greenback cutthroat trout (*O. c. stomias*) by Cummings (1987) indicated that juvenile brook trout excluded juvenile cutthroat trout from more favorable stream positions.

A population of WCT inhabits White's Creek, a tributary to the upper Missouri River entering Canyon Ferry Reservoir south of Helena, Montana (Fig. 1). By 1993 WCT in White's Creek had been reduced to about 80 individuals that occupied two different areas of the stream separated by an intermittent segment of channel that only flowed during high flow events. Factors believed primarily responsible for this low WCT abundance included the invasion and establishment of a relatively strong brook trout population and impacts of past placer and dredge mining activities on aquatic habitats over a 1 km portion of the stream channel below the intermittent channel. Moore et al. (1983), Moore et al. (1986), Larson et al. (1986), and Kulp and Moore (2000) found that repeated, intensive electrofishing removals conducted over time reduced or exterminated nonnative rainbow trout populations allowing native brook trout populations to rebound in streams of the Great Smokey Mountains National Park. Thompson and Rahel (1996) evaluated depletion electrofishing for removal of brook trout in three streams of Wyoming to conserve Colorado River cutthroat trout and found that densities of brook trout could be dramatically reduced, but not eradicated, by three-pass depletion electrofishing. They also reported that an additional single electrofishing pass conducted the year after the three-pass effort helped to further reduce brook trout numbers, especially age-1 fish that were missed as age-0 the previous year.

We wished to evaluate if 1) repeated electrofishing efforts could successfully

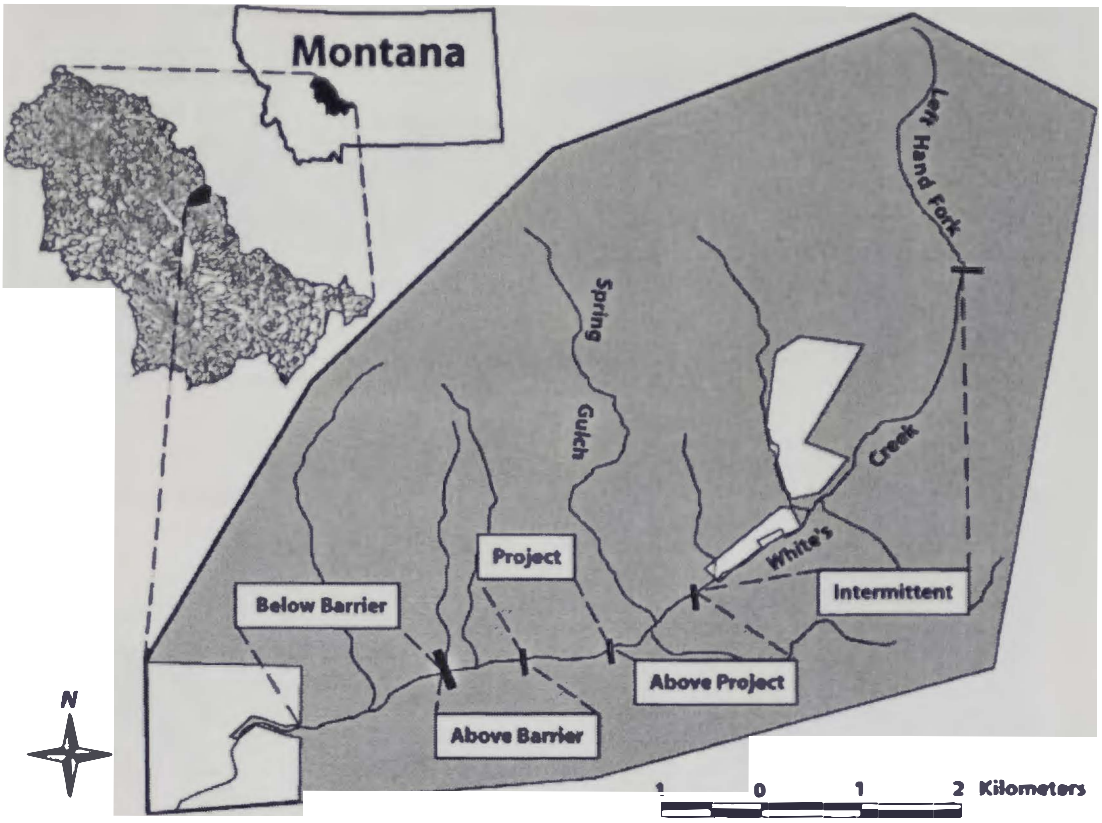

Figure 1. Map of White's Creek study area showning Forest Service (shaded) and private (unshaded) ownership, location of constructed fish barrier (dark bar), and reaches that were sampled (boxes designate named reaches).

remove brook trout from White's Creek, and 2) reductions or elimination of brook trout would result in positive population responses by WCT and, if so, what demographic parameters of the WCT population would be affected. In addition, we discuss whether combined habitat reclamation and brook trout removal would result in a different response by WCT than brook trout removal alone; and if interactive mechanisms between brook trout and WCT could be inferred from the population response exhibited by WCT following removal of brook trout.

Study Area Description

White's Creek lies within the White's Gulch drainage on the east side of Canyon Ferry Reservoir, a large reservoir on the Missouri River above Helena, Montana (Fig. 1). All White's Creek's flow from the Forest Service boundary (stream km 8.8) downstream was diverted for flood and sprinkler irrigation, except during extreme high spring runoff events. This de-watering of the channel below the Forest Service boundary has isolated the upper portion of the drainage from Canyon Ferry Reservoir. Flows were also intermittent in the upper basin, from stream km 13.6 up to the Left Hand Fork of White's Creek (km 18.4; Fig. 1). Base summer flows in White's Creek below Spring Gulch Creek usually ranged from 0.05 to 0.15 m^3/sec and there was high inter-annual variation in summer monthly flows during this study (Fig. 2). Wetted widths averaged about 2 m, average depth was about 10 cm, pools comprised 10-20 percent of all habitat types, and while small (<150 mm diameter) woody debris was fairly common (75-175 pieces/km), large (>150 mm) debris was relatively scarce (4-19 pieces/km) based on habitat surveys in three separate sections of approximately 100 m. About 400 m of channel in the Left Hand Fork of White's Creek maintained an

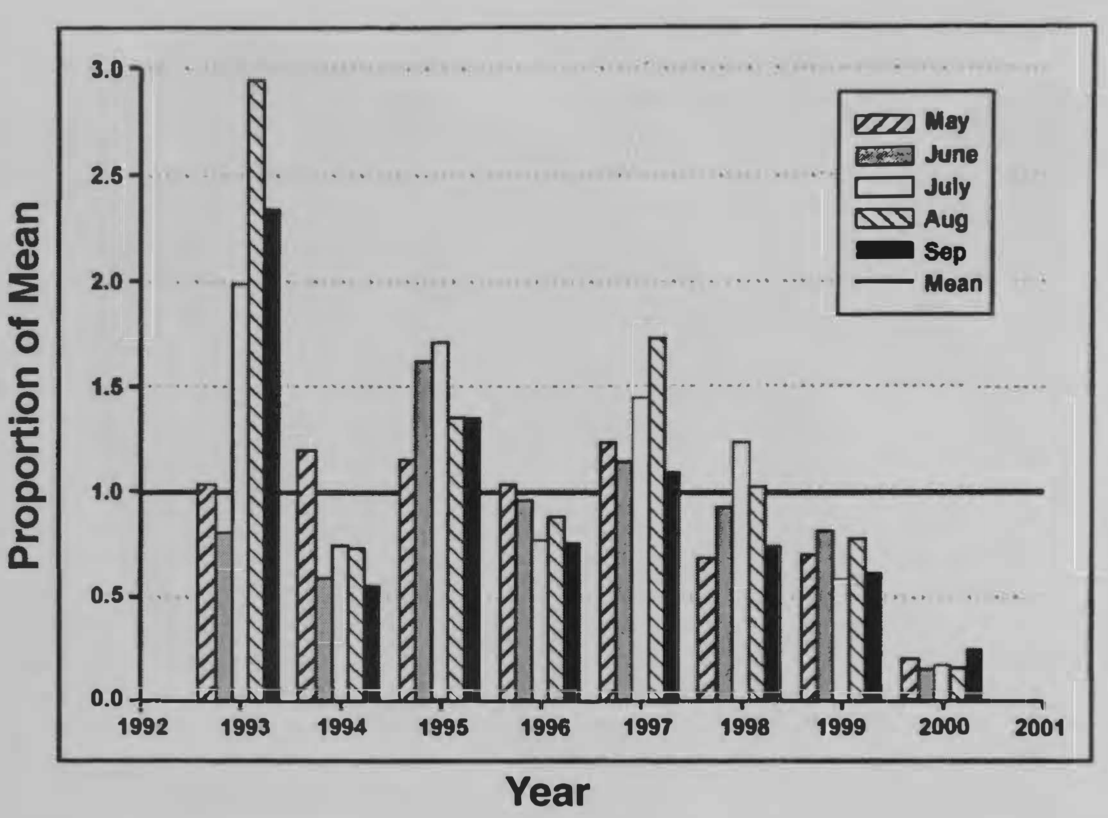

Figure 2. Proportion of mean monthly flows from May through September from 1993 through 2000 in Prickley Pear Creek, a USGS gauged tributary near White's Creek, based on 59 years of record showing the exteme high flows of summer 1993, higher than average flows in 1995 and 1997, and drought conditions of 1999 and 2000.

estimated yearlong flow (~0.03 m^3/sec) originating from springs, but its flow subsided into the streambed near its mouth with White's Creek. Onset Optic Stowaway® thermographs deployed during the summers of 1997 and 1999 at two locations below Spring Gulch Creek measured average daily summer water temperatures that ranged from 8 to 10 °C.

Past placer and dredge mining activities heavily impacted the portion of White's Creek from stream km 11.4 to 12.4. These impacts included the diversion of most of White's Creek's flow into a diversion canal located along the north valley wall, several large dredge ponds located within the valley floor, and intermittent flows between some of these dredge ponds (Fig. 3, top). An extremely high flood event occurred after an intense local thunderstorm during the summer of 1993 (Fig. 2). This flood event resulted in breaching of the diversion canal and several of the dredge ponds creating several large headcuts within the channel.

The only fish species found in White's Creek above the Forest Service boundary were WCT and brook trout. No rainbow trout have been captured in White's Creek above the Forest Service boundary although rainbow trout from Canyon Ferry Reservoir have been sampled spawning in the lowermost reaches of White's Creek immediately above the reservoir.

Methods

During the fall and winter of 1995 reclamation of the mining impacted reach (1.0 km; Project reach) of White's Creek was undertaken. This reclamation included total reconstruction of the stream channel and valley bottom; installation of 11 grade control structures across the channel, eight of which crossed the entire valley width; placement of woody debris in the channel

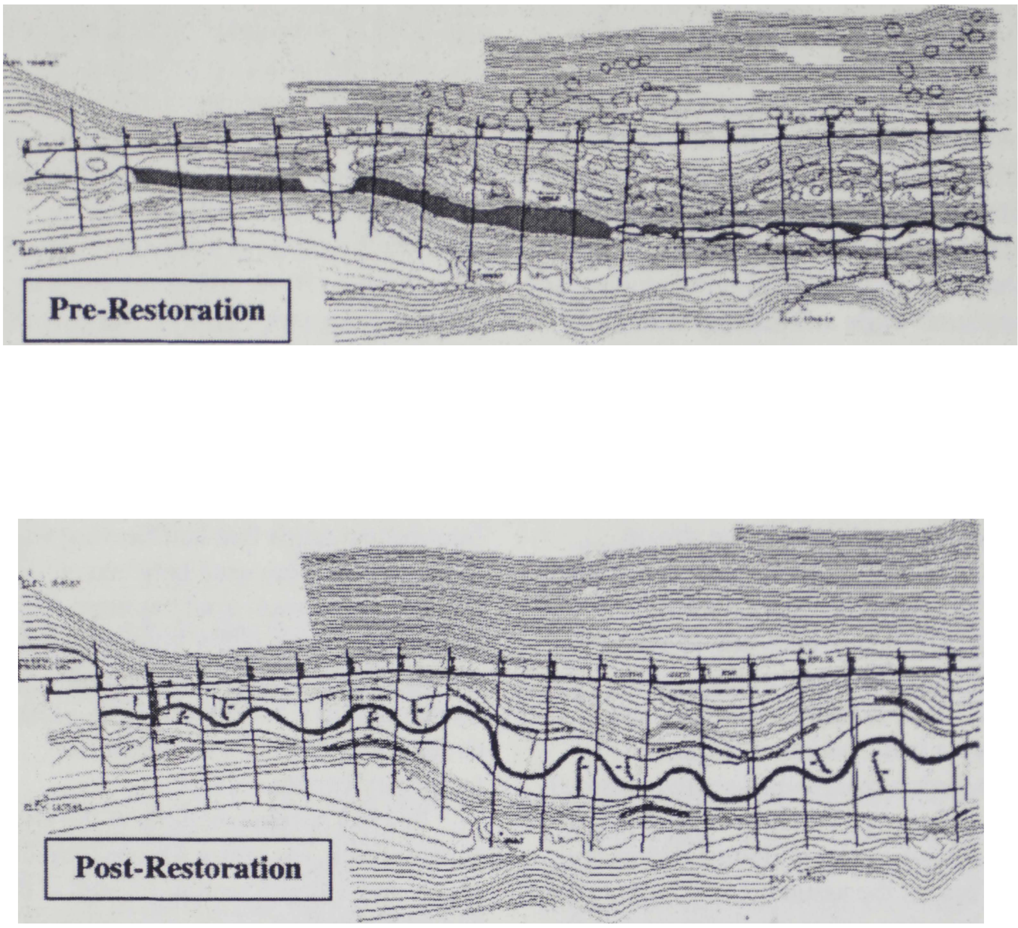

Figure 3. Plano-metric maps of the Project reach of White's Creek pre- and post-restoration showing contours, channel location, and cross-sections (maps provided by Inter-fluve, Bozeman, Montana).

and on the floodplain; and planting of vegetation throughout the floodplain (Fig. 3). A barrier to upstream fish movement also was constructed at stream km 10.6 in the fall of 1995. This barrier was constructed using treated wood and consisted of a 1.5-m drop onto a treated wood apron that is designed to prevent the formation of a scour pool at the base of the barrier that could be used as a jump pool. In addition, the channel was widened immediately below the barrier to prevent water from backing up against the barrier at peak streamflows.

Four designated reaches were surveyed during the study. The first reach (Below Barrier) went from the Forest Service boundary upstream to the constructed barrier. This reach was about 1800 m long, but most sampling occurred immediately below the constructured barrier. The second reach (Above Barrier) extended from the wooden crib barrier up to the bottom end of the reclaimed portion of the stream and was approximately 750 m long. The third reach (Project Area) consisted of the entire 1000-m long reclaimed portion of the channel. The fourth reach (Above Project) was from the top boundary of the Project upstream 1200 m to where the flow in the channel normally becomes intermittent. The Left Hand Fork (Fig. 1) may have contributed a few downstream WCT migrants to the study reaches during high water flows; however, the population in the Left Fork was very low and stable during the study (~30 WCT ≥75 mm long).

Electrofishing was used to remove brook trout and estimate populations of brook trout and WCT using depletion estimators (Van Deventer and Platts 1989). Fish were captured using Smith-Root BP-15 and BP-12 backpack electrofishers operated at voltages in the range of 100-600 V, frequencies under 50 Hz, and pulse widths less than 2 msec to maximize the number of fish captured while minimizing injury to fish caused by the shock (Dwyer et al. 2001). An electrofishing crew consisted of a crewmember wearing the backpack shocker, a primary dip netter that followed the shocker, and usually a backstop netter who kept a large dip net in the stream channel below the two other crewmembers. The backstop net was large enough that it generally spanned at least half of the channel in most sample sections. In addition, block fences or nets (6.5-mm mesh) were installed between sample sections during most sampling and removal events. All electrofishing passes in each sample section were conducted within four hours. We met the assumption of population closure by using either block fences or nets at the upper and lower ends of all sample sections (or, in a few cases, locating sections so they had shallow riffles or velocity barriers at their upper and lower boundaries), the use of a back-stop netter during sampling to prevent fish from moving downstream, and the relatively short time it took to complete all sample passes (White et al. 1982).

Total lengths (mm), species, and pass number were recorded for all captured fish. Weights (g) were not measured for all captured fish, but a relatively large sub-sample of fish, evenly distributed among all the fish sampled, were weighed during initial brook trout removal efforts and during at least two of the post-removal sampling efforts. For a few sample events neither lengths nor weights were measured for brook trout <100 mm (age-0), instead they were counted and their total numbers were recorded by pass and species. Age-0 (<60 mm) WCT were usually not netted during sampling to reduce handling mortality; however, relative abundances of age-0 WCT were noted on data sheets based on observations during sampling.

During the summer of 1993, prior to the major flood event, depletion population estimates were made in several sample sections in the Above-Project reach; however, brook trout were not removed during these efforts. After the summer 1993 flood event, a temporary barrier culvert was installed between the Above-Project and Project reaches at about stream km 12.4. Multiple electrofishing passes (usually two) were made throughout the entire Above-Project reach and a portion of the Intermittent reach that had flowing water. Block fences were used between all sample sections. All brook trout captured during these efforts were either killed and buried on-site (fish <150 mm), killed and transported to a Food Bank (fish ≥150 mm), or relocated downstream below the temporary barrier culvert.

In 1994 several sections in the Below Barrier reach, the entire Above-Project reach, and a single section located at the top end of the Project reach were sampled. All brook trout captured in the Above Project reach were killed and buried on-site. In 1995 the dredge ponds were sampled by a raft-mounted electrofishing unit while being drained in preparation for reclamation construction. Due to high numbers of brook trout captured in the dredge ponds by electrofishing and netted during the draining of these ponds, neither an exact count nor any measurements of brook trout removed from these ponds was recorded. Instead, the total number of brook trout removed was estimated by counting the approximate number of brook trout in a hand held net full of fish and counting the number of nets full of fish removed from these ponds. The entire Above-Barrier reach was sampled twice in 1995, following construction of the barrier and removing brush in this portion of the stream channel. Brush removal consisted of cutting and removing vegetation and woody debris that overhung the channel and a few accumulations of debris within the channel

Table 1. Length sampled and total numbers of brook trout removed from three reaches of White's Creek above a constructed fish barrier from 1993 through 2001. The number of brook trout removed from the Project reach during 1995 was estimated based on footnote.

	Above-Barrier		Project		Above-Project		
Year	Length (m)	EBT removed	Length (m)	EBT removed	Length (m)	EBT removed	Total EBT removed
1993	0		130		1200[1]	111	111
1994	0		230		830	33	33
1995	750	1499	1000	2750[2]	1200	22	4271
1996	750	138	800	4	350	2	144
1997	750	119	1000	12	350	4	135
1998	750	59	1000	9	1000	170	238
1999	750	60	800	4	1130	27	91
2000	750	4	400	0	620	0	4
2001	750	0	50	0	0		
Total		1879		2779		369	5027

[1] A second sampling was done over 1,000 m of this reach later in the year and the recorded number of brook trout removed was for both sampling events.

[2] Number approximated based on approximate number of fish per dip net and total number of dip nets full of fish removed.

to permit easier access by shocking crews. In addition, the entire Above-Project reach was sampled in 1995. Following construction of the wooden barrier at stream km 10.4, the temporary barrier culvert between the Project and Above-Project reaches (km 12.4) was removed. All brook trout captured during 1995 were relocated below the wooden barrier. A few WCT captured within the Project area in 1995, immediately prior to reclamation, were relocated to locations either immediately above the wooden barrier or above the project area.

Electrofishing estimates and associated brook trout removal efforts continued annually from 1996 through 2000 in the Above-Barrier, Project, and Above-Project reaches; however, in 1996 and 1997 only three monitoring sections (each 100 to 150 m long) within the Above-Project reach were sampled (Table 1). Prior to shocking the Above-Barrier reach in 2000 brush was again cleared to permit shocking crews easier access.

Indices of relative abundance were derived by reporting catch of each species (fish $\geq$ 75 mm) on the first electrofishing pass, standardized as number/100 m of channel length for all sampling events. We used depletion estimators to calculate population estimates (Van Deventer and Platts 1989) for fish $\geq$75 mm long and converted to density of fish/ha for all multiple-pass sampling events. Depletion estimators consistently under-estimate true populations, especially when only two passes are made and capture probabilities are <0.90 (Riley and Fausch 1992). White et al. (1982) recommended that three or more passes are necessary unless the capture probability is $\geq$0.8. Riley and Fausch (1992) suggested that three passes reduced estimate bias. Of the 52 estimates of WCT we made, 41 were two-pass estimates and 11 were three-pass estimates. Estimated probabilities of capture were at least 0.7 for all two-pass population estimates for fish 75 mm and longer and were $\geq$0.8 for 80 percent of the population estimates.

We estimated total biomass by species for three sample sections in the Above-Project reach by averaging weights of all captured fish by species for length groups of 75-149 mm and $\geq$150 mm. Average weights and estimated numbers for each size group were used to estimate standing crop for each size group, as well as total standing crop, and reported as grams/m^2; however, we did not include age-0 WCT in total standing crop estimates. Condition factors for individual fish were computed as Fulton-type condition factors (Anderson and Gutreuter 1983) and averaged by reach for all fish ≥75 mm long captured during late summer to early fall (Aug-early Oct). We used a statistical *t*-test assuming unequal variances to test for significant differences ($P < 0.05$ indicated significance) in fish condition by species between fish captured before and during initial brook trout removal efforts (before removal) versus those captured following initial brook trout removal efforts (after removal).

Results

Brook Trout Removal

Slightly over 5000 brook trout were removed from the portion of White's Creek above the wooden barrier from 1993 to 2000 (Table 1). Most of these (~4200) were removed in 1995 when the dredge ponds were drained. Once removal efforts began in each reach, numbers of brook trout removed in subsequent years declined dramatically. However, we thought extensive removal efforts conducted in the Above-Project reach from 1993 to 1995 were successful, so we did not attempt extensive removal efforts in 1996 and 1997. Instead, we limited our efforts to monitoring three sample sections of 100-150 m each to document recovery of WCT. Apparently brook trout either successfully re-colonized this upper portion of the creek from lower reaches, or some mature brook trout evaded removal efforts and successfully reproduced, because in 1998 we captured numerous brook trout in this reach. Consequently, during 1998 we again removed brook trout from this entire reach. By year 2000 we captured only four brook trout while sampling all reaches despite extensive sampling efforts and found no brook trout in 2001 in an 800-m section above the wooden barrier (all of the Below-Project reach and about 50 m of the Project reach).

We more effectively removed larger brook trout (>100 mm) than smaller fish, particularly during early removal efforts (Fig. 4). For example, only brook trout ≥130 mm were captured in initial removal efforts in the Above-Project reach during 1993. During 1994 a few adult-sized and numerous age-0 brook trout were captured in this reach, and by 1997 and 1998, only age-0 and age-1 brook trout were captured. After these age-0 brook trout reached age-1 (>100 mm) in 1998 and 1999 they were more vulnerable to capture.

Average condition factors of brook trout were significantly ($P<0.001$) lower after we began to remove them (1.16 before, $n=237$, versus 1.04 after, $n=204$). Average condition factors still significantly differed ($P<0.001$) when assessed for fish ≥100 mm long (1.17 before removal, $n=177$, and 1.09 after, $n=97$), which suggested that though we more effectively removed larger brook trout during our initial removal efforts, this was not the only cause for differences in condition factors.

Response of Westslope Cutthroat Trout

Numbers of WCT increased following removals of brook trout; however, numbers of cutthroat trout increased only slightly during the first two years following the initial brook trout removal effort. By the fourth year following initial brook trout removals, populations and relative abundance of WCT had reached levels similar to, or higher than, combined populations of brook and cutthroat trout when removals began (Fig. 5 and 6). The estimated population of WCT 75 mm and longer increased almost seven-fold in the Above-Project reach from 1993 to 1999; from about 80 fish to over 340. The increase was even more dramatic in the Above-Barrier reach, where the estimated population increased from about 10 in 1995 to over 340 in 1999.

Estimates of standing crops (g/m^2) in three sample sections within the Above-Project reach showed that, following an

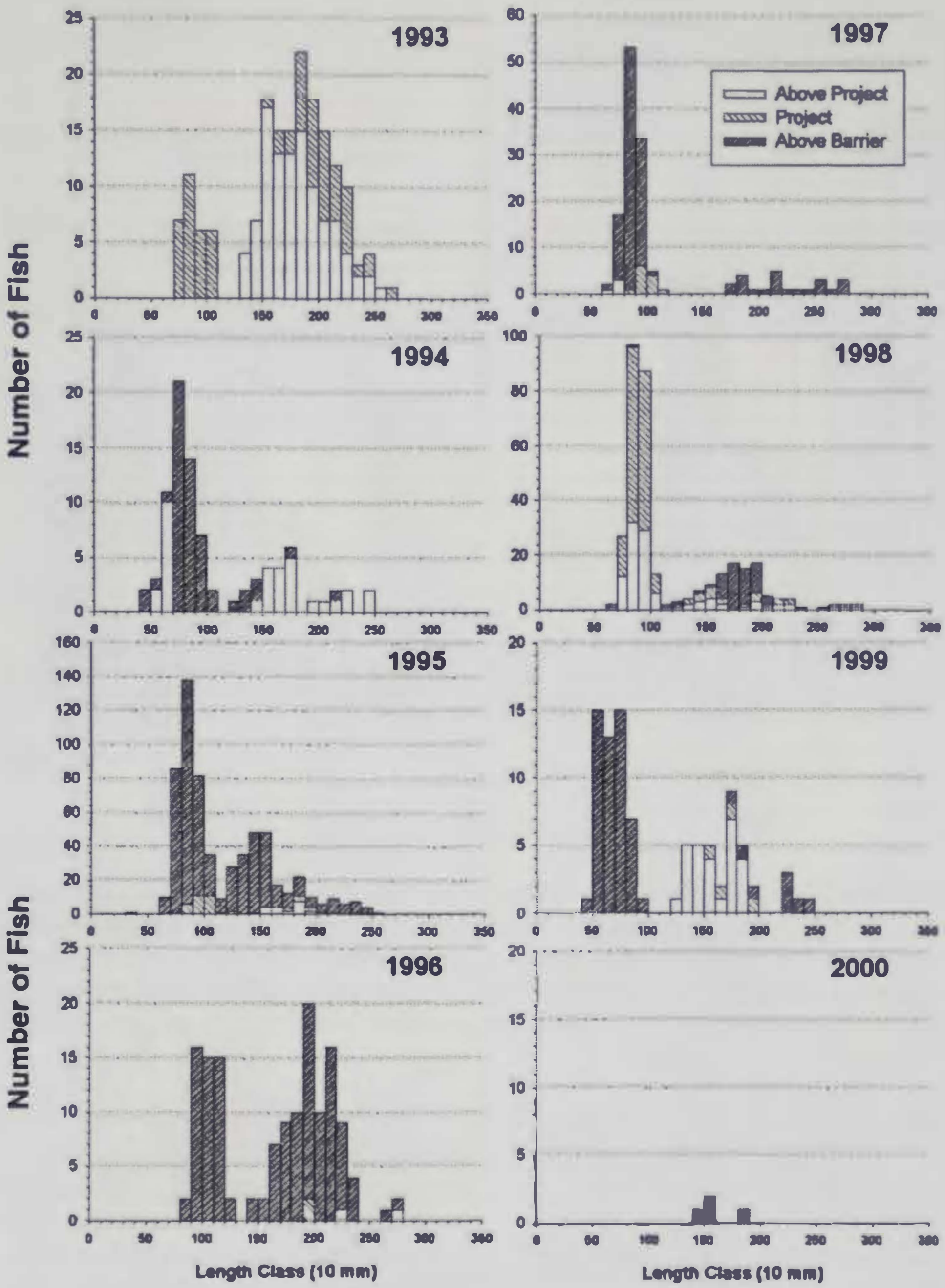

Figure 4. Length frequency histograms for brook trout captured and removed from three reaches of White's Creek from 1993 through 2000. Note that y-axis scales change and that the number of small (<100 mm) brook trout captured increased, while the number of larger brook trout decreased after initial removal efforts.

initial decline in standing crops immediately following the first two years of brook trout removals, WCT standing crops increased to levels at least as high as those observed when both species existed in sympatry (Fig. 7). Standing crop data showed less yearly fluctuations than population and relative abundance data and standing crop information is probably a better measure of population responses because it is a more consistent and stable measure of abundance. Average condition factors of WCT (≥75 mm) were slightly, but not significantly (P=0.14), lower following

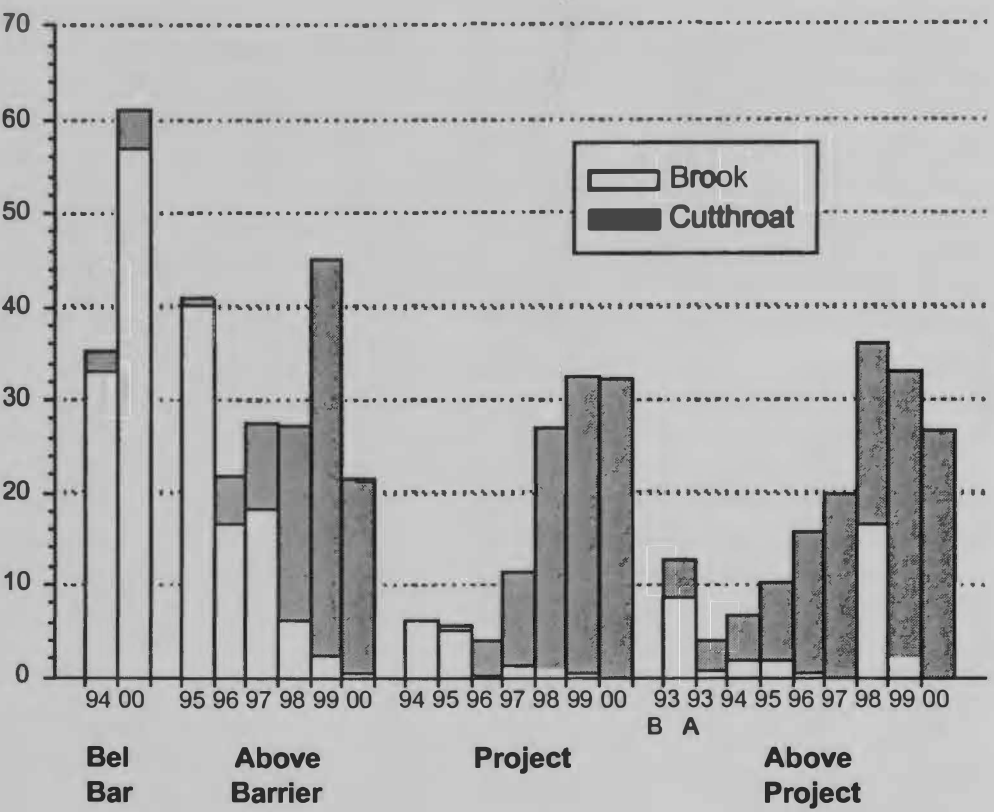

Figure 5. Relative abundance (catch per 100 m in the first electrofishing pass) of brook trout (open bars) and westslope cutthroat (shaded bars) 75 mm and longer in four reaches of White's Creek from 1993 to 2000. Brook trout were not removed from the reach below the wooden crib barrier (Bel Bar). Some of the Above Project reach was sampled once before brook trout were removed in 1993 (93 B) and immediately after brook trout were removed (93 A).

removal of brook trout (0.96 before removal, n=151, versus 0.94 after, n=1497), but were not different when assessed for WCT ≥100 mm long (average condition of 0.94 both before removal, n=133, and after, n=1008; P=0.94).

Length frequency histograms and observations of age-0 WCT that were seen, but not captured, indicated that age-0 WCT (<60 mm) comprised progressively larger proportions of the population after 1995 (Fig. 8). Numerous age-0 WCT were observed, but no attempts were made to capture these age-0 within the Project reach in 1996, one year following its reclamation. In 1997, when efforts were made to capture small WCT, age-0 WCT fish made up a significant proportion of captured fish in all three treated reaches and this strong 1997 year-class of WCT carried through as age-1 (~80-100 mm) fish in 1998 (Fig. 8). Approximately 30 percent of all captured fish ≥75 mm long were longer than 200 mm in the Above-Project reach during 1993 when brook trout removals began. The proportion of these ≥200 mm fish increased to 34-55 percent during 1997 through 1999; however, the proportion of larger fish declined in 2000.

Habitat Restoration

Our observations indicated that flows during peak flow events remained within the constructed Project reach channel from 1995 to 2001, despite higher than normal peak flows experienced during 1997 (Fig. 2). We observed some over-bank flows for a short time during 1996 and 1997, but erosion of valley surface materials outside

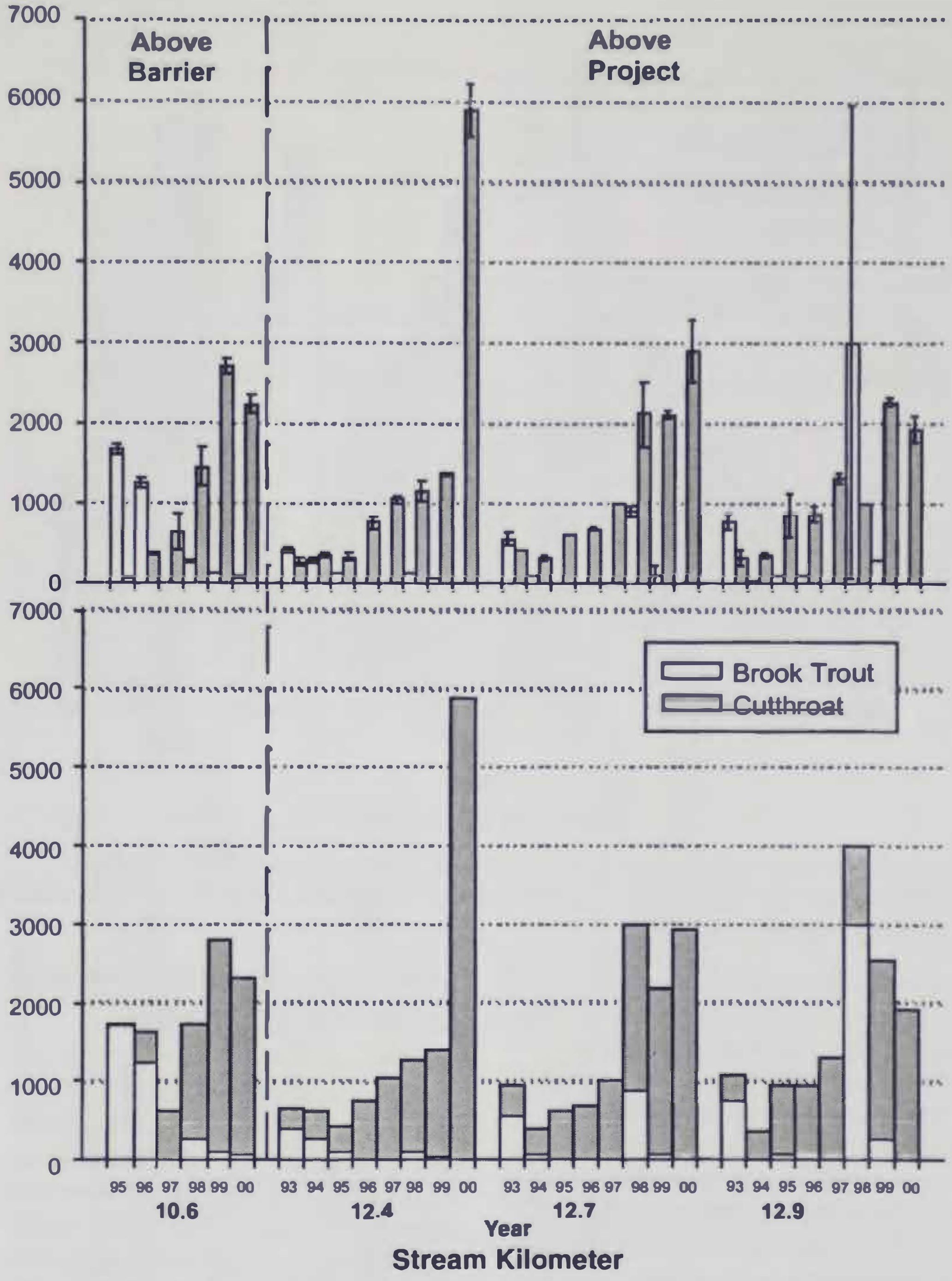

Figure 6. Estimated numbers of brook (open bars) and westslope cutthroat (shaded bars) 75 mm and longer per hectare of stream surface area and associated 95% confidence intervals (vertical lines-top graph) in four sections (designated by stream kilometer) of White's Creek from 1993 through 2000. Kilometer 10.6 was within the Above Barrier reach and km 12.4, 12.7, and 12.9 were within the Above Project reach.

the stream channel was minimal; no new channels were created during peak flow events. We believe that incorporation of underlying valley-wide grade control structures prevented high flows from eroding new stream channels. Valley bottom hay bales and silt fences also effectively prevented rill and channel erosion down the valley floor during the four years it took for valley-bottom vegetation to re-establish. Woody species have yet to become well established, despite two attempts to plant woody species sprigs.

Surface stream flows often become

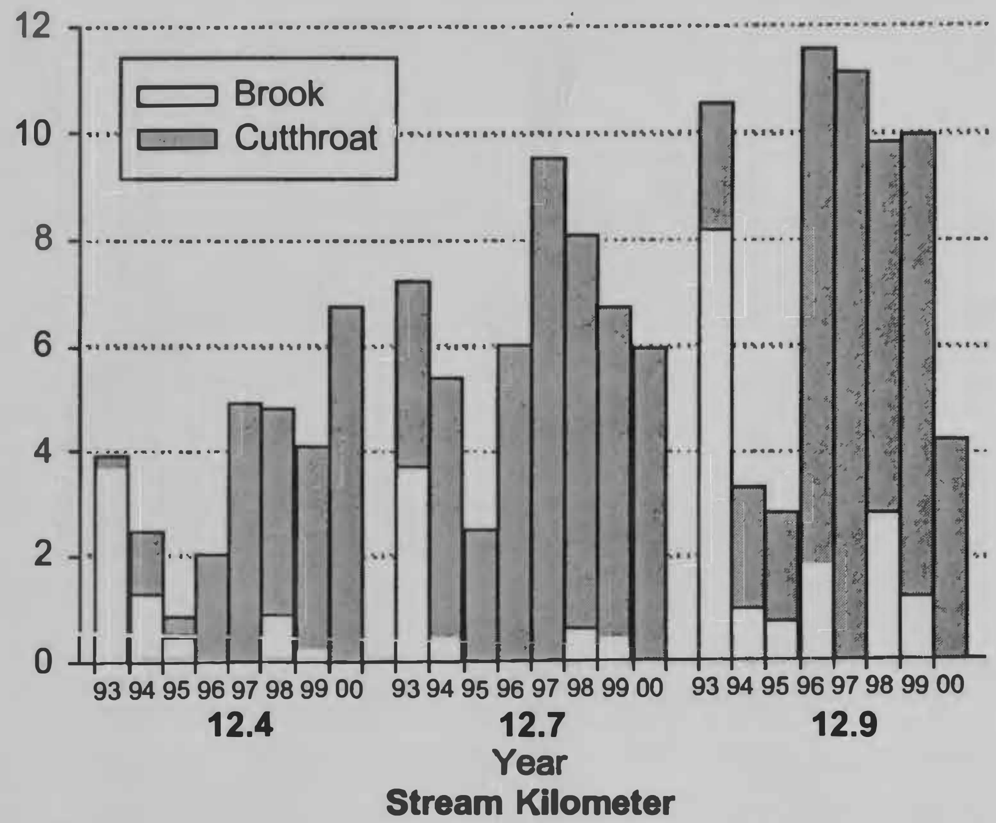

Figure 7. Estimated standing crops (g/m^2) of brook trout (open bars) and westslope cutthroat trout (shaded bars) in three sample sections of White's Creek located in the Above Project reach from 1993 to 2000.

intermittent in the lower portion of the Project reach during base flow periods. The length of intermittent channel declined annually as fine sediments carried by high flow events helped seal the streambed; however, drought conditions in 1999 and 2000 have contributed to the lower portion of this restored channel going dry (Fig. 2). Woody debris elements and excavated pools incorporated into the channel also maintained their integrity throughout this period.

Average daily water temperatures from July 15 to September 15 averaged 2.6 °C higher below the Project reach than above during the summer of 1997, immediately following construction. However, by 1999, after riparian vegetation had begun to establish, differences in average daily water temperatures above and below the Project Area were much less pronounced, averaging only 0.9 °C higher below the project area.

Effect of Habitat Restoration on Abundance of Westslope Cutthroat Trout

Relative abundances of WCT ≥75 mm long within the three reaches from 1998 through 2000 indicated that abundances of WCT within the Project reach were similar to the Above-Barrier and Above-Project reaches (Fig. 5). These data suggest that channel restoration completed in the Project reach provided habitat that was as suitable for WCT as habitats provided by the natural channel immediately above and below the Project reach in White's Creek. Re-colonization of this Project reach by WCT, primarily age-0, began immediately following restoration and removal of all fish from this reach, and progressed rapidly so that this Project reach supported densities of WCT similar to adjacent reaches within about three years.

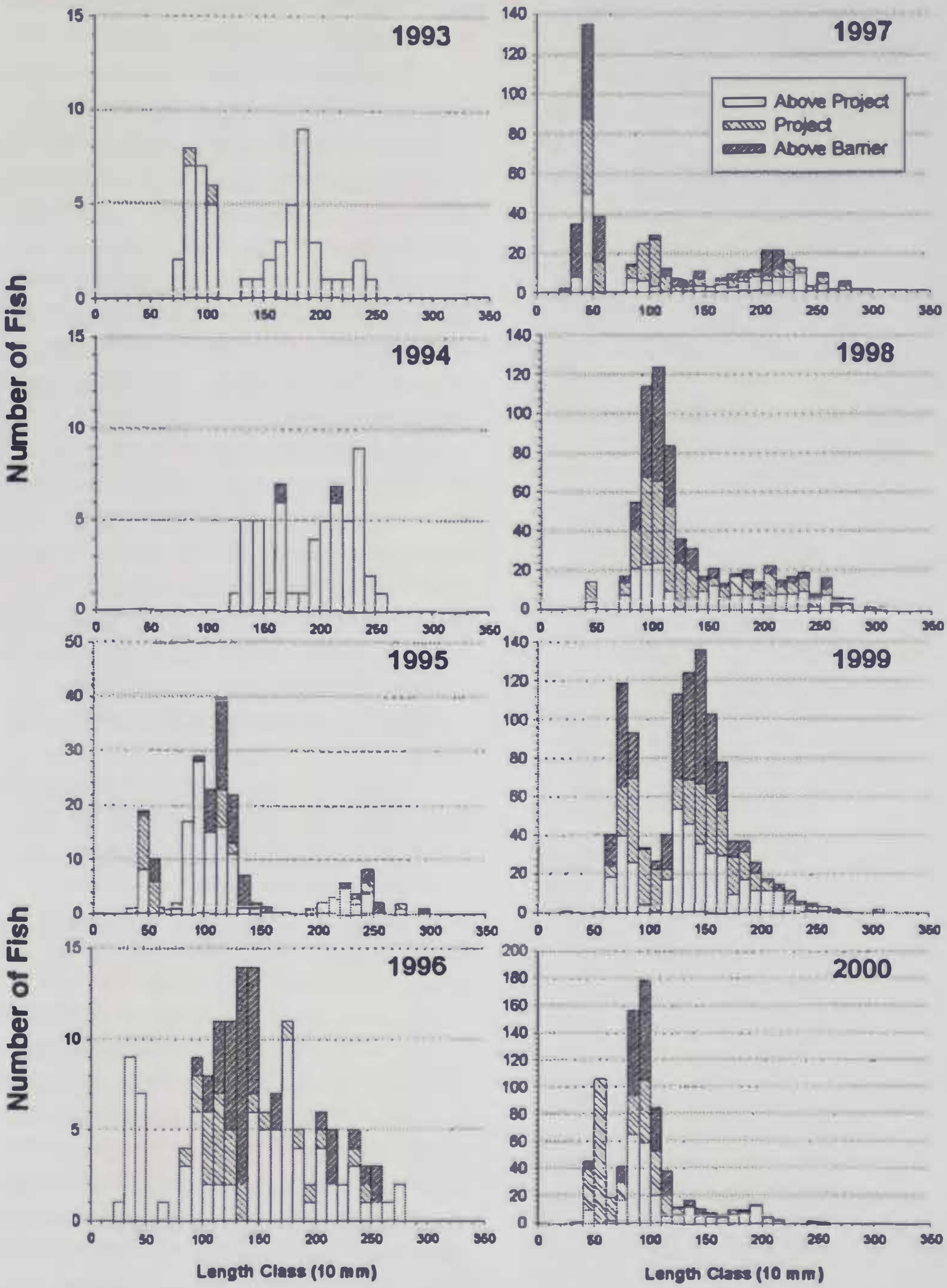

Figure 8. Length frequency histograms for westslope cutthroat trout captured in three reaches of White's Creek from 1993 through 2000. Note that y-axis scales change and that number of smaller age-0 (< 60 mm) and age-1 (60 to 120 mm) westslope cutthroat trout captured increased dramatically following initial brook trout removals (1993 through 1995).

Discussion

Effectiveness of Brook Trout Removals using Electrofishing

In this small, simple stream, we effectively removed brook trout using electrofishing; however, it took us eight years to accomplish an apparent total removal. Sampling found no brook trout in the Project and Above-Project reaches during 2000 and no brook trout in the Above-Barrier reach during 2001.

Thompson and Rahel (1996) found that single three-pass electrofishing removal efforts in headwater portions of three Wyoming tributaries that supported native Colorado River cutthroat trout successfully removed 59-100 percent of the age-1 and older brook trout, based on depletion efficiencies, and over 90 percent based on number of fish removed. They found that efficiencies for removal of age-0 brook trout were lower, based on number of fish removed (42-83%). Moore et al. (1983), Moore et al. (1986), and Kulp and Moore (2000) found that repeated intensive electrofishing over time was effective at reducing or removing nonnative rainbow trout populations from streams in Great Smokey Mountains National Park that historically supported native brook trout populations. Kulp and Moore (2000) found that four three-pass electrofishing efforts within the same year effectively eliminated recruitment of rainbow trout, but a fifth effort was necessary to totally eliminate rainbow trout.

We enhanced our removal efficiency by having vegetation and small woody debris that overhung the stream channel trimmed twice during the seven-year brook trout removal effort to allow our shocking crews easier access to the channel. De-watering and reclamation of the dredge pond areas of stream allowed us to totally remove brook trout from these areas of the stream, something that would have been impossible to do by electrofishing these ponds when they were full of water.

Repeated removal over the entire stream was necessary for at least four consecutive years, similar to results reported by Kulp and Moore (2000). We mistakenly believed we had effectively removed brook trout from the Above Project reach after three years of removals, so we only conducted limited removal efforts in association with annual monitoring of three sample sections during 1996 and 1997. Brook trout populations expanded in this reach during that time period and additional intensive removal efforts were required during 1998 and 1999 to effectively eliminate brook trout from this reach.

We found that initial removal efforts were more effective at removing larger (>100 mm) brook trout. Once most of these larger trout had been removed, we could concentrate more of our effort to remove smaller brook trout. This finding was consistent with results reported by Kulp and Moore (2000) for removal of rainbow trout and Thompson and Rahel (1996) for removal of brook trout. Following brook trout removal efforts in 1995 and 1996, it appeared that only the 1998 year-class of brook trout (spawned in the fall of 1997) was very successful. Consequently, once these 1998 year-class fish reached a size where they were more vulnerable to backpack electrofishing (>100 mm) during 1999, they were effectively removed. No successful brook trout reproduction appeared to occur above the barrier after 1998.

A barrier to upstream fish movement is necessary at the lower boundary of any removal project to ensure that nonnative species do not move upstream to re-colonize reclaimed habitats. Existing cutthroat trout recovery plans recognized the importance of barriers to prevent competition and hybridization with nonnative trout species (USDI Fish and Wildlife Service 1993a, 1993b, Langlois et al. 1994). Harig et al. (2000) found that many of the greenback cutthroat trout restoration attempts that failed were due to competition with nonnative salmonids. In most cases, removal efforts were not totally effective, but in some cases re-invasion over man-made barriers occurred following the restoration. Harig et al. (2000) cautioned that man-made barriers are not as effective as natural waterfalls. Thompson and Rahel (1998) evaluated fish passage at man-made barriers thought to be protecting native Colorado River cutthroat trout from invasion by brook trout. They found several brook trout moved upstream past a rock-gabion barrier and one brook trout was found above a culvert barrier, but they speculated that an angler moved this trout.

Response of Westslope Cutthroat Trout to Suppression of Brook Trout

Westslope cutthroat trout populations increased dramatically after three years of brook trout removal, and the most dramatic increases were observed in abundances of age-0 WCT soon after removal efforts began. Cummings (1987) and Thomas (1996) indicated that juvenile brook trout interfered with juvenile cutthroat trout's foraging efficiency and microhabitat selection. Numerous age-0 WCT utilized the recently re-constructed channel within the Project Area immediately after its construction, probably due to vacant habitats it provided. The standing crop of WCT in allopatry was similar to the combined standing crop of WCT and brook trout in sympatry; however, condition of WCT did not significantly change following brook trout removal. Research in Idaho indicated that removal of brook trout prior to stocking greatly enhanced stocking success for westslope cutthroat trout fry (Cowley 1987, Strach and Bjornn 1989).

Confounding Effects of Habitat Restoration and Brook Trout Removal

Our data did not show that WCT populations within the Project reach were any higher than in the other two reaches where brook trout were also removed, but no habitat enhancement was undertaken. Thus, we could not test if, or how much, habitat restoration contributed to the population level of WCT we observed in the Project reach. In addition, our data did not let us empirically test whether habitat restoration alone would have resulted in a community shift favoring WCT; however, we suggest that habitat restoration alone probably would not have resulted in much of a response by the WCT population without simultaneous removal of brook trout. We reached this conclusion because prior to their removal, brook trout dominated the fish community in the two reaches adjacent to the dredge pond reach, where stream habitats were in relatively good condition. In these adjacent reaches WCT were being replaced by brook trout, even in relatively high-quality stream habitats. The combination of habitat restoration and brook trout removal within the Project reach resulted in densities and standing crops of WCT similar to adjacent reaches where stream habitat was in good condition. Thus, all we can conclude is that habitat restoration successfully provided habitats of similar quality as present in adjacent natural reaches. Habitat restoration has been shown to increase abundance and distribution of cutthroat trout (House and Boehne 1985 and 1986, Young et al. 1999); however, in these studies cutthroat trout either existed in allopatry or in sympatry with other native fishes and thus, did not face competition from nonnative salmonids.

We acknowledge that these dredge pond habitats could have provided a source of brook trout to adjacent habitats while allowing brook trout to dominate fish communities in these adjacent reaches. However, research we have conducted in other streams, where no similar ponds existed (either beaver or dredge), suggests that brook trout will replace WCT, even in the absence of pond area sources (Shepard et al. 1998). Pond habitats also might have raised stream temperatures below these ponds, affecting interactions between brook trout and cutthroat trout to favor brook trout (DeSato and Rahel 1994); however, this mechanism is speculative because we have no water temperature information from below the ponds prior to their removal.

Fausch (1989) suggested that distributions of brook trout and WCT might be influenced by stream gradient, with brook trout occupying lower gradient stream reaches (with maximum abundance observed at gradients <3%), and WCT occupying higher gradient reaches (with maximum abundance in gradients ranging from 6 to 14%). The senior author and others previously developed a regression model that related the presence and abundance of brook trout and numerous habitat variables to explain observed variation in abundance of WCT (Shepard et al. 1998). Using principal components analyses, they derived components indicative of water temperature, pool frequency, stream size, stream gradient, and ranked levels of logging, road construction, livestock grazing, and mining activities that, along with brook trout presence and abundance covariates, led to regression equations that significantly (R^2 from 0.79 to 0.80; P<0.01) related to observed abundance of WCT. Based on these studies and our observations in White's Creek, we suggest that in streams with lower stream gradients and warmer water temperatures, such as White's Creek, brook trout populations will likely replace WCT populations. To conserve WCT in these types of streams, brook trout will have to be removed or periodically suppressed.

Native Fish Conservation Management Implications

White's Creek was initially chosen to test the feasibility of physically removing nonnative brook trout and assess the response of the native WCT population to that removal. While removal of the brook trout population was accomplished, it took seven years of intensive effort. The WCT population responded positively, increasing from an initial population of less than 75 fish (ages 1 and older) to over 1000 age-1 and older fish; however, the low initial population size of WCT raises concerns for potential genetic inbreeding depression and a severe population bottleneck (Meffe 1986, Allendorf and Leary 1986). We observed some WCT with malformed opercules, but are unsure if this was related to genetic problems or diet. We suggest that restoration efforts should be initiated before populations decline to levels below 50 effective breeding individuals (e.g., Allendorf et al. 1997), which probably translates to at least 100 adult-sized (>130 mm; Downs et al. 1997) fish.

Our results suggest that brook trout replace WCT and that the mechanism for this replacement may be behavioral interaction between age-0 brook and WCT. Following removal of brook trout abundances of age-0 WCT increased dramatically in all reaches, but especially within the Project reach where all fish had been removed. Since brook trout emerge several months earlier than WCT (early summer versus late summer) they have a competitive size advantage over WCT in the same cohort that is maintained through at least their first year of life. Griffith (1972) found that age-0 brook trout maintained a 20-mm size advantage over WCT of the same age group in Idaho streams and consistently dominated age-0 WCT during behavioral interactions; however, he believed interactions were minimized due to utilization of different microhabitats. Sabo and Pauley (1997) suggested that size is perhaps equally important as species in determining competitive dominance between sympatric populations of cutthroat trout and coho salmon (*O. kisutch*). We suggest that most streams have limited microhabitats consisting of slow, shallow-water habitats near cover for age-0 trout; consequently, intense behavioral interactions likely occur between age-0 brook trout and WCT with brook trout dominating due to their larger size.

We recognized that successful restoration of WCT requires sites containing high quality habitats (Griffith et al. 1989) in a mosaic that will change over time (Young 1995). Ideally these sites should include refugia (Sedell et al.1990, Pearsons et al. 1992) where some individuals could withstand extreme events and subsequently disperse to re-colonize vacant habitats. Hilderbrand and Kershner (2000) suggested that total habitat size should support 2500 individuals, based on earlier work by Allendorf et al. (1997), which they translated to stream lengths of about 10 to 50 km depending upon relative fish abundance. Harig et al. (2000) evaluated success of 37 greenback cutthroat trout (*O. c. stomias*) restoration efforts and found that 23 had failed due to reinvasion of nonnative species (48% of the failures) or because restoration was done in unsuitable habitats (43%).

A potential problem associated with long-term persistence of the WCT in White's Creek is the fact that this WCT population has a relatively short reach of available habitat, about 3 km, that puts this population at a high risk of extinction due to both stochastic and demographic pressures (Rieman and McIntyre 1993, 1995, Hilderbrand and Kershner 2000, Harig et al. 2000). Hilderbrand and Kershner (2000) recommended a population of at least 2500 individuals to avoid inbreeding depression and reduce extinction risk. Using our catch rates of about 30 fish/100 m (Fig. 5) along with an estimated capture efficiency of about 0.8 expanded over the 3 km of available habitat results in a ballpark total estimate of about 1125 WCT in the portion of White's Creek above the barrier. This suggests that the WCT population in White's Creek is at a relatively high risk of going extinct, a fact we readily acknowledge. Fortunately, White's Creek has several perennial springs that enter White's Creek within the restoration area, providing a more stable environment and local refugia that might reduce extinction risk from stochastic environmental events.

We acknowledge the relatively high extinction risk for WCT in White's Creek, even after the elimination of brook trout due to their limited numbers and restricted and isolated habitat. However, we suggest that genetic introgression and nonnative competition threats may outweigh stochastic risks over the short-term, making isolation of many remaining WCT populations a reasonable and necessary short-term conservation strategy. Montana's conservation agreement for WCT (Montana Fish, Wildlife and Parks 1999) calls for replication and re-founding of existing populations that will likely be lost due to stochastic or demographic pressures. Montana fish managers have recognized that human intervention will be necessary to act as the dispersal agent to re-found WCT populations lost from isolated headwater habitats due to stochastic processes.

While physical removal of brook trout was ultimately possible in White's Creek, we do not believe it is a viable alternative in most places currently occupied by sympatric populations of WCT and nonnative competitors. White's Creek is a small (2-m wetted width and average depth of ~10 cm), relatively uncomplex stream with little woody debris and relatively low levels of instream cover, making shocking efficiencies relatively high. In addition, the restoration within the Project reach allowed for total removal of brook trout from in-channel ponds. We believe that chemical removal of nonnative fish will be necessary in most systems (Stevens and Rosenlund 1986, Gresswell 1991, Buktenica et al. 2000, Brooks and Propst 2001).

Another consideration relates to the cost of physical removal. We estimate that it cost at least $30,000 for the eight years of removals, assuming an hourly salary and benefits rate of $10, daily per diem rates of $37, and associated travel costs (all costs are US 1999). It cost an additional $15,000 for the barrier installation. We did not include costs of the valley restoration, since this restoration was a mining reclamation effort; however, this restoration work contributed to the removal effort by draining ponds occupied primarily by brook trout. These costs must be viewed as low because this stream was easily accessible throughout its length and was located relatively close to a field office. This translates to about $10,000/km for removal treatments plus the cost of the barrier.

In conclusion, this study indicated that 1) brook trout could successfully be removed from a small, relatively simple stream using electrofishing, but it took eight years of effort to accomplish, 2) a severely depressed WCT population re-bounded to levels at least seven times higher than pre-removal estimates following brook trout removal, 3) biomass of WCT in allopatry was similar to the combined biomass of brook trout and WCT in sympatry, and 4) where habitat restoration occurred in conjunction with brook trout removal, densities and biomass of WCT were nearly as high as adjacent natural sections of stream channel.

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