Conserving Wild Trout

WILD TROUT SYMPOSIUM

Holiday Inn Resort
West Yellowstone, MT
September 28-30, 2010

Wild Trout X

Carline, R.F., and C. LoSapio, editors. 2010. Conserving wild trout. Proceedings of the Wild Trout X symposium, Bozeman, Montana. 370 pages. (Copy available at www.wildtroutsymposium.com).

WILD TROUT X STEERING COMMITTEE

Dirk Miller, Symposium Chair
Liz Mamer, Secretary
Joe Urbani, Treasurer
Kevin Meyer and Doug Besler, Program Committee
Amy Wolfe, Fundraising
Rob Gipson, Facility Committee
Leah Elwell and Marty Seldon, Publicity
Robert Carline, Proceedings Editor
Carol LoSapio, Proceedings Publisher
Jim Daley, Awards Committee
Kathy Buchner, Jim Stelfox, and Jennifer Earle Organizing Committee

Organizing Committee: (front row) Jennifer Earle, Amy Wolfe, Chris Kozafky, Carol LoSapio, Joe Urbani, Bob Carline, Marty Seldon, Kathy Buchner; (middle row) Mark Novak, Rob Gipson, Dirk Miller, Liz Mamer, Bob Gresswell, Doug Besler, Scott Stanton, Tracey Stephens; (back row) Kevin Meyer, Bob Wiltshire; (missing Committee members: Jim Daley, Jim Stelfox, and Leah Elwell).

To order additional copies of these proceedings contact:
Joseph Urbani & Associates, 1502 Gold Avenue, Bozeman, MT 59715, 406-587-0588.

CD's with additional information/files are available at no cost. CD contents: Proceedings of Wild Trout X Symposium, photos of the activities and events, acceptance award speeches, awards brochure, WTX program, and PowerPoint shows.

Information is also available at: www.wildtroutsymposium.com

Logo design by Zach Matthews, American Angler Editor, The Itinerant Angler (www.itinerantangler.com)

PROCEEDINGS OF THE WILD TROUT X SYMPOSIUM
WEST YELLOWSTONE, MT
SEPTEMBER 28-39, 2010

SYMPOSIUM CHAIR, DIRK MILLER

TECHNICAL EDITOR, BOB CARLINE

SYMPOSIUM PROCEEDINGS PUBLISHER, CAROL LOSAPIO

A REMINDER . . . WTXI WILL BE HELD IN 2014.
STAY IN CONTACT THROUGH: WWW.WILDTROUTSYMPOSIUM.COM

TABLE OF CONTENTS

Wild Trout X Symposium Program

2010 Program Schedule

Monday, September 27, 2010

1:00	Free fly-casting clinic by Bob Jacklin
4:00	Wild Trout X Steering Committee Meeting
5:00	Registration Opens/Speaker Presentation Check-in
6:00	Wild Trout Ice Breaker Social

Tuesday, September 28, 2010

7:00	Breakfast/Registration Desk Open/ Speaker Presentation Check-in
8:00	Morning Session
12:00	Lunch
1:00	Afternoon Session
4:30	Poster Session
5:30	Adjourn *(Dinner on your own or join us at 7:00 for "heavy hors d'oeuvres" and cash bar before and during the movie.)*
7:00	Movie: "Rivers of a Lost Coast"

Wednesday, September 29, 2010

7:00	Breakfast/Registration Desk Open/ Speaker Presentation Check-in
8:00	Morning Session
11:50	Awards Luncheon
1:00	Afternoon Session
4:30	Poster Session
5:30	Adjourn
6:30	Wild Trout Banquet

Thursday, September 30, 2010

7:00	Breakfast
8:00	Morning Session
12:10	WTX Wrap-up Session
1:00	Post-symposium Debrief and WTXI Planning *(Box Lunch reservations required for interested volunteers and Committee members. All welcome to attend!)*

Fly Casting Clinic by Bob Jacklin

Local fishing celebrity Bob Jacklin shares fly casting techniques and fishing magic with all.

SPONSORS

The Wild Trout Symposium gratefully appreciates the support provided by these agencies and individuals. These contributions help preserve, protect and perpetuate wild trout around the world for the generations to come. For more information concerning sponsorship opportunities, visit us at www.wildtroutsymposium.com.

Premier Sponsors: .... US Fish & Wildlife Service
Trout Unlimited

Major Sponsors: .... Madison River Foundation
Bureau of Land Management

Event Sponsors: .... USDA Forest Service
Jackson Hole One Fly - National Fish & Wildlife Foundation
Urbani Fisheries, LLC
Fisheries Conservation Foundation

In Kind Sponsors:......................... Federation of Fly Fishers
Trout Ball by Greg Keeler
Performance Fly Rods

Awards: .... Aldo Starker Leopold Wild Trout Awards
Advanced Telemetry Systems Inc.
Ron Remmick Undergraduate Student Scholarship Award
Marty Seldon Graduate Student Scholarship Awards

Premier Sponsors

US Fish & Wildlife Service

The U.S. Fish and Wildlife Service is a bureau within the Department of the Interior. Our mission is to work with others to conserve, protect and enhance fish, wildlife and plants and their habitats for the continuing benefit of the American people. Objectives: 1) Assist in the development and application of an environmental stewardship ethic for our society, based on ecological principles, scientific knowledge of fish and wildlife, and a sense of moral responsibility 2) Guide the conservation, development, and management of the Nation's fish and wildlife resources; and 3) Administer a national program to provide the public opportunities to understand, appreciate, and wisely use fish and wildlife resources.

Trout Unlimited

Today TU is a national organization with more than 150,000 volunteers organized into about 400 chapters from Maine to Montana to Alaska. This dedicated grassroots army is matched by a respected staff of lawyers, policy experts and scientists, who work out of more than 30 offices nationwide. These conservation professionals ensure that TU is at the forefront of fisheries restoration work at the local, state and national levels.

Nearly 50 years after its founding, no other conservation organization is as well placed as TU to make a difference for the nation's coldwater fisheries. To learn more about TU's ambitious conservation agenda, please visit the conservation section of our website TU Conservation.

Major Sponsors

Madison River Foundation

Founded in 2003, the Madison River Foundation is an advocate for the Madison amid the challenges of rapid residential and population growth, commercial development, increasing recreational use and the traditional Western competition over scarce water resources. We strive to work collaboratively with all those who live, work, and recreate on this storied river and its related watershed.

Based in Ennis, Montana, the Madison River Foundation is a tax-exempt, non-profit membership organization incorporated under Section 501(c)(3) of the Internal Revenue Code. We are supported by the voluntary contributions of members, friends and donors.

The Foundation has a dual mission: advocacy and conservation. In its advocacy role the Foundation seeks to be “a voice for the river” in the public arena, advocating worthy public and regulatory policies based on sound science. In its conservation role, the Foundation funds and provides through its membership “boots in the water” volunteer labor for a variety of conservation-oriented projects.

Bureau of Land Management

The BLM manages more inland fish habitat than any other State or Federal agency, including 117,000 miles of fish-bearing streams and over 3 million acres of lakes and reservoirs. The more than 250 million surface acres managed by the BLM contain diverse water bodies, from isolated desert springs harboring populations of rare and unique fish to large Columbia River tributaries that provide habitat for Pacific salmon and steelhead as they migrate long distances to breed.

BLM waters support subsistence fisheries that sustain traditional Native American cultural heritages, as well as nationally significant recreational fisheries such as Gunnison Gorge in Colorado, Lake Havasu in Arizona/California and the Rogue River in Oregon. Public lands also support 127 federally-listed threatened or endangered aquatic species, 155 BLM sensitive species, and a variety of highly valued sport fish species. In short, BLM's rivers, lakes and streams are of great ecological, cultural, and recreational importance.

Event Sponsors

USDA Forest Service

The USDA Forest Service has nine regions that encompass over 200,000 miles of perennial stream and over 2 million acres of lake and reservoir habitat. Wild trout and their habitat are a key component of National Forest System lands. The Forest Service has been a leader in working with partners to conserve coldwater habitat and in supporting recovery and conservation of native and desired non-native trout fisheries. The variety of trout and salmon habitat on National Forests ranges from the highest elevation mountain streams to coastal ecosystems on the west and east coasts of the United States, with key inland habitats that support many of the endemic North American native trout species. The Forest Service is proud to be a sponsor of Wild Trout X.

Jackson Hole One Fly - National Fish & Wildlife Foundation

The Jackson Hole One Fly-National Fish and Wildlife Foundation Conservation Partnership invests in projects and initiatives that help protect and restore native trout populations and their habitats in the intermountain west. For information on grant programs and projects funded, contact Cara Rose, NFWF at cara.rose@nfwf.org

Urbani Fisheries, LLC

Urbani Fisheries, LLC specializes in the creation and restoration of stream, wetland, and lake ecosystems. Involved in fisheries enhancement and management for over 30 years, Joe Urbani has successfully integrated experienced professionals who combine their diverse expertise and knowledge into a results-oriented team.

Urbani Fisheries is a recognized provider of aquatic habitat enhancement and reconstruction of stream, wetland and lake ecosystems that have been degraded by past land and stream management practices that altered or degraded habitat. Our team maintains an efficient, cost-effective and common sense approach throughout the life of a project while embracing a strong commitment to the environment. The company has successfully completed numerous projects across the country resulting in the creation, restoration and enhancement of warm and cold water fisheries, waterfowl, and wildlife habitat.

Recently, Urbani Fisheries has been involved in channel restoration, stream bed manipulation, and habitat enhancement. Most recent involvement has been with Heritage brook trout in the Mosconetcong River and wild brown trout in the Pecos River. In-stream enhancements, combined with bank stabilization and riparian restoration techniques, have been implemented on multiple sections of the Musconetcong River in New Jersey and the Pecos River in New Mexico.

Urbani Fisheries prides itself on making sure all projects emphasize aesthetic appeal in conjunction with the creation of biologically functioning, ecologically diverse, natural systems.

Fisheries Conservation Foundation

Founded in 2003, the Fisheries Conservation Foundation (FCF) is a nonprofit organization that supports the work and knowledge of aquatic scientists, resource managers, and environmental professionals. Our mission is to promote a better understanding of marine and freshwater fishery resources among fishery users, the general public, and political decision-makers, and to encourage the enlightened management of fisheries resources for their optimum use and enjoyment by the public.

The FCF strives to ensure that objective, peer-reviewed scientific information about fisheries and aquatic resources reaches policy-makers and the public, so the decisions made about the use of our freshwater and marine ecosystems are logical, informed, and based on the principles of sustainability.

To be effective in creating solutions for today's complex aquatic resource problems, the Fisheries Conservation Foundation works to form partnerships with other conservation organizations that have similar goals. By joining with our strategic partners, we collectively work to make a real difference.

In-Kind Sponsors

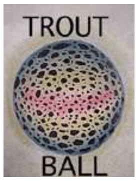
Federation of Fly Fishers

The Federation of Fly Fishers is a 43 year old international non-profit organization dedicated to the betterment of the sport of fly fishing through Conservation, Restoration and Education. The Federation of Fly Fishers and its Councils are the only organized advocate for fly fishers on an national and regional level.

The idea to create a national federation of fly fishing clubs by people who also were willing to take action, seems to have sprung up on both coasts in the early 1960's. In June 1965, the first Conclave of the Federation of Fly Fishers took place in Eugene Oregon and was hosted by the McKenzie Flyfishers. Today the FFF has grown to over 300 clubs, and the organization is moving more and more toward being an organization comprised of individual members. Our goal is to support fisheries conservation and educational programs for all fish and all waters. Anywhere fly fishers have an interest, the FFF can and does play a role.

The FFF joined Trout Unlimited in becoming a cosponsor of the Wild Trout symposiums at WT-II in 1978. Through dedicated staff and continued support, the mission of Wild Trout has benefited from this generous vision. Many thanks to the individuals involved in making this happen.

Trout Ball by Greg Keeler



Greg Keeler generously crafted the song "Born to be Wild" for the WT-IX Symposium in 2007, in West Yellowstone, Montana. Professor Keeler teaches English by day at Montana State University-Bozeman and entertains the rest of us with wonderful, irreverent, original songs concerning all things fishing. Take a moment to visit his website for captivating art and prose. Thank you, Dr. Keeler, for sharing your gifts with the Wild Trout Symposium.

Performance Fly Rods

Dave Lewis, a lover of nature and visual arts, as well as stellar custom rod maker, spent many an hour capturing the beauty of fish and fishing across the country. The wild trout community lost Dave to cancer in 2008, after a long full life of appreciating the spirituality of wild trout country. His photos grace the WT Symposium web pages and we thank him for the opportunity to experience these extraordinary images and through them, him.

Aldo Starker Leopold Wild Trout Award Sponsor

Advanced Telemetry Systems Inc.

Advanced Telemetry Systems, Inc. is an innovative, science and engineering-based radio telemetry provider, dedicated to supporting biologists worldwide. Our commitment to our customer's success has helped us build a reputation as the leader in fisheries and wildlife research. We've partnered with preeminent researchers to design the most reliable radio tracking systems ever deployed. The experienced professionals at ATS possess a thorough understanding of the challenges you'll face in the field, and we're ready to provide you complete solutions - and valuable customer support - for your study's radio tracking equipment needs.

The Wild Trout Symposium gratefully acknowledges ATS and their support of the Aldo Starker Leopold Medal. Dick Reichle and his wife, Laura, stepped forward to make the casting of these bronze medals possible. Thank you.

Awards

Committee Members

Jim Daley, Chariman	Robert Behnke
Gardner Grant	Robert L. Hunt
Ron Jones	Bud Lilly
Nick Lyons	Nathaniel P. Reed
Frank Richardson	Marty Seldon
R.P. (Van) Van Gytenbeek	Ray J. White

The Wild Trout Symposium brings together a broad and diverse audience of governmental entities, non-profit conservation groups, media representatives, educators, anglers, fishing guides, and business interests associated with trout fisheries to exchange technical information and viewpoints on wild trout management and related public policy. Held every 3 years, each symposium has led to innovative wild trout management approaches.

The following awards were made at Wild Trout X:

- Aldo Starker Leopold Wild Trout Medal - Professional
- Aldo Starker Leopold Wild Trout Medal - Non-Professional
- Ron Remmick Undergraduate Student Scholarship Award
- Marty Seldon Graduate Student Scholarship Awards (two will be awarded)
- Trout Unlimited-Federation of Fly Fishers Wild Trout Stewardship Award

Aldo Starker Leopold Wild Trout Medal

The Wild Trout Symposium Organizing Committee established the Aldo Starker Leopold Wild Trout Medal in 1984 as a continuing memorial to this distinguished naturalist, teacher, author, and an important participant in these symposia, who was the son of Aldo Leopold. Two Wild Trout medals are conferred, one to a professional and one to a nonprofessional individual, who in the eyes of their peers have made long-time and significant contributions to the enhancement, protection, and preservation of wild trout.

Ron Remmick Undergraduate Student Scholarship Award

This award will recognize one outstanding undergraduate student with a strong interest in conservation and restoration of native trout, and is offered to encourage their participation in Wild Trout Symposia. This award consists of a $400 stipend to assist student travel or other costs incurred in attendance of this symposium. This award is open to undergraduate students in Fisheries Management or related field. Applicants will be judged on a combination of an essay written by the applicant and a letter of reference. An application form can be found at the conference website at www.wildtroutsymposium.com.

Marty Seldon Graduate Student Scholarship Award

In order to recognize outstanding students in the field of fisheries management and biology, and to encourage their participation in Wild Trout Symposia, two students will be awarded with Marty Seldon Student Scholarships at each symposium. Each award consists of a $500 stipend to assist student travel or other costs incurred in their attendance. Awards are open to graduate students in Fisheries Management or related field. Applicants will be judged on a combination of current graduate GPA and an essay written by the applicant. An application form can be found at the conference website.

TUFFF Wild Trout Stewardship Award

This award is conferred by the Wild Trout Symposium for the implementation of an outstanding fishery project or plan that makes a significant contribution to the conservation, protection, restoration, or enhancement of a cold water fishery. This award is made to a club, group, or other organization. The Symposium Awards Committee reviews all nominations and selects appropriate recipients. The physical award consists of a certificate of achievement and an honorarium of $1,000 underwritten equally by the Federation of Fly Fishers and Trout Unlimited.

WILD TROUT X SYMPOSIUM AWARD RECIPIENTS

Charles Cathcart

Ron Remmick Undergraduate Student Award

Charles Cathcart

In recognition of an outstanding undergraduate student with a demonstrated interest in conservation and restoration of native trout.

Daniel A. James

Marty Seldon Graduate Student Award

Daniel James

In recognition of an outstanding graduate student in the field of fisheries management and biology.

Bradly Allen Trumbo

Marty Seldon Graduate Student Award

Bradly Trumbo

In recognition of an outstanding graduate student in the field of fisheries management and biology.

Little River Chapter TU

Trout Unlimited/Federation of Fly Fishers Wild Trout Stewardship Award

Little River Chapter TU

For significant contributions to the conservation, protection, restoration, or enhancement of a coldwater fishery.

Thomas R. Pero

Wild Trout Medal

Nonprofessional Award

Thomas Pero

To a non-fisheries professional who, in the eyes of their peers, have made long-time and significant contributions to the enhancement, protection, and preservation of wild trout.

Stephen E. Moore

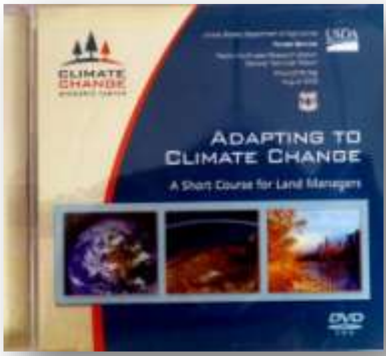
Wild Trout Medal

Professional Award

Stephen Moore

To a fisheries professional who, in the eyes of their peers, have made long-time and significant contributions to the enhancement, protection, and preservation of wild trout.

Symposium Events & Activities



Adapting to Climate Change DVD

The Michael J. Furniss, US Forest Service Pacific Northwest Research Station, produced short course "Adapting to Climate Change" DVD has an excellent section on fishery impacts. Copies were made available to WT-X Symposium attendees at registration. Additional copies may be obtained by requesting USFS Pub No: PNW-GTR-789 from: 503-261-1211, pnw_pnwpubs@fs.fed.us, http://www.fs.fed.us/ccrc/hjar/

Wild River Press Donates Books

Wild Trout X would like to thank Tom Pero's Wild River Press for donating 6 cartons of books to attendees!

Movie — Rivers of a Lost Coast

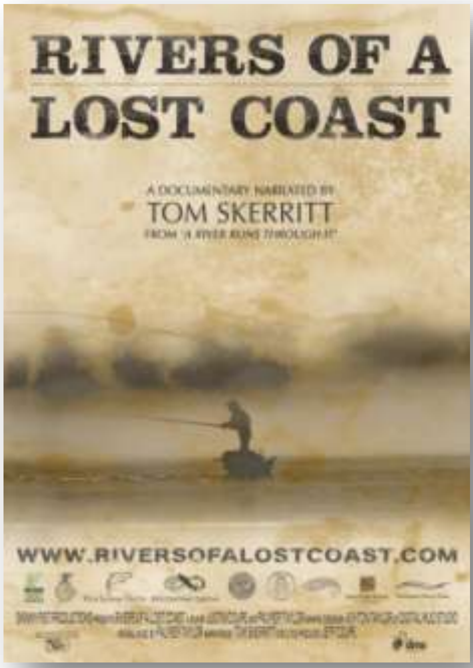
As an added feature for the symposium this year, participants were invited to view a documentary narrated by Tom Skerritt. For more information visit: http://www.riversofalostcoast.com/index.php

Story

At the turn of the 20th Century, a handful of pioneers carried their fly rods into California's remote north coast and gave birth to a culture that would revolutionize their sport. For a select few, steelhead fly fishing became an obsessive pursuit without compromise.

Leading the pack was the mythical, Bill Schaadt, an off-kilter angler famous for his ruthless pursuit to be 'in the fish'. The new endeavor was ruled by a demanding, unspoken code, which made 'breaking in' almost as difficult as 'breaking out'.

By the early 1980s, the Golden State's coastal fisheries found themselves caught in a spiraling decline. As California searched for its disappearing salmon and steelhead, these men foraged for their souls.



Plenary Session

PEOPLE AND TROUT: IMPLICATIONS OF SOCIAL AND ECONOMIC TRENDS FOR WILD TROUT AND ASSOCIATED HABITATS

Barbara A. Knuth

Human Dimensions Research Unit, Department of Natural Resources, Cornell University, Ithaca, New York.

Examining social and economic trends associated with freshwater fishing and trout fishing specifically can yield useful insights for managers. The National Survey of Fishing, Hunting, and Wildlife-associated Recreation (U.S. Fish and Wildlife Service, http://wsfrprograms.fws.gov/subpages/NationalSurvey/National_Survey.htm), provides national, regional, and state-level data about every 5 years that can be used to understand trends in fishing participation, economic impact, and characteristics of anglers. Focused statewide angler surveys can provide additional information about in-state fishing participation trends among residents and nonresidents, relative use of different types of water bodies and preferred species, and insights about anglers' management preferences. Mass-media sources reflect interests and values among anglers and other groups within the general public. This presentation includes a summary of trends from the National Survey, key findings regarding trout-related fishing participation, and angler management preferences from two statewide angler survey efforts in New York and Vermont, and two mass-media examples related to environmental attitudes and trout fishing.

The 2006 National Survey reports 27% of U.S. freshwater anglers fished for trout. Trout fishing participation differed by region of residence, with the largest number of trout anglers living in the Pacific, Mountain, and Middle Atlantic regions. The number of anglers participating in trout fishing has declined from 1996 to 2006. The states with the highest trout fishing participation have shifted slightly over time, with California, Pennsylvania, and Colorado as the top three in both 1996 and 2006, but Washington shifting from 4th-most to 5th-most and New York from 5th-most to 4th-most anglers in that same time period. Total trout fishing days have also declined from 1996 to 2006, but California and Pennsylvania were first and second-highest states for trout fishing days in both time periods. Average days spent per angler trout fishing have remained steady over time, at about 11 days per angler. Based on 2006 data, women comprise a smaller percentage of U.S. trout anglers (21%) than U.S. freshwater anglers (25%). Age of trout anglers has shifted. In 1996, the 25-34 and 35-44 age groups dominated among trout anglers, but in 2006, the dominant age groups were 35-44 and 45-54. Trout anglers tend to be more educated than the general U.S. population, with more trout anglers having completed college or advanced higher education. Trout anglers tend to have a higher household income than the U.S. population and freshwater anglers as a group. In 2006, the National Survey reports that 6.8M trout anglers spent US$4.8B on fishing-related expenses, creating a $13.6B economic impact and supporting over 100K jobs nationwide.

Vermont statewide angler surveys conducted in 1991, 2000, and 2010 provide insights regarding trout angling trends. Angler participation in fishing for trout in small brooks or beaver ponds and in large streams or rivers is declining, while participation for bass fishing is increasing. The mean number of days spent fishing for trout has remained about the same over this time period, but has increased for bass fishing. Anglers appear to have shifted in their support for specific management activities over time, with both residents and nonresidents becoming more supportive of stocking trout in some streams and rivers. Although nonresident angler support for managing some streams and rivers only for wild trout increased over time, resident angler support did not increase. Angler support for special regulations for trout fishing on some waters has increased over time, for regulations dealing with lower creel limits, catch and release fishing, and use of artificial lures and flies. Perceptions about the acceptable "keeper" and "quality" size trout have changed over time, with the smallest "keeper" size fish increasing for brook trout, but the smallest "quality" size fish decreasing or remaining the same for brook trout.

New York statewide angler data from 2007 provide an example of how managers and human dimensions researchers partnered to identify typologies of angler clusters, leading to managers identifying appropriate management responses for each angler cluster. These included a cluster focused on coldwater species; a cluster focused on fishing inland lakes, including Adirondack ponds, for trout plus warm water species; and a cluster focused on fishing inland streams for trout and warmwater species. These clusters differ somewhat in their fishery management preferences, such as the importance of wild fish, satisfaction with number and size of fish caught, and interest in opportunities to catch wild trout.

Fishery managers can consider these trends and these angler classification approaches in terms of implications for adapting to a declining constituent base, responding to an increasing interest in active management and specialized fishing regulations, anticipating implications of climate change and associated species shifts, and understanding and managing angler expectations.

.

TRUTH, LIES, AND MYTHS IN THE AGE OF INSTANTANEOUS INFORMATION: REDEFINING THE ROLES OF ANGLERS AND FISHERIES PROFESSIONALS IN WILD TROUT MANAGEMENT

Steve L. McMullin

Department of Fish and Wildlife Conservation, Virginia Tech, Blacksburg, VA 24061, **smcmulli@vt.edu**, (540) 231-8847

Effective agencies maintain close and open relationships as they involve their stakeholders in decision-making processes, while simultaneously balancing biological concerns with public opinion effectively. Performing that balancing act, always a difficult task, has become even more difficult as the many avenues of communication and the speed with which stakeholders communicate among themselves and with fisheries managers have changed. In this paper, I discuss the evolution of processes for how fisheries professionals have involved stakeholders in planning and decision-making processes. I also discuss how the proliferation of methods of electronic communication among stakeholders and between stakeholders and fisheries professionals has altered those relationships. Finally, I reflect on the implications of information sharing today and in the future on management of wild trout.

Throughout my career, I have approached planning and decision making for fisheries and wildlife management with an egalitarian philosophy of fisheries management based on the notion that stakeholders and professionals should have equally important but distinct roles in the decision-making process. Implementation of that philosophy involves focusing stakeholder participation on making value choices that define management goals and focusing the participation of professionals on providing technical information about the implications of management choices to achieve goals.

Implementation of the values choices/technical choices philosophy in Montana led to considerable success in developing biologically sound fisheries management plans that also had broad stakeholder support. Evaluation of a management planning process for black bears in Virginia, based on the values choices/technical choices philosophy, demonstrated that stakeholders who participated in the process became more knowledgeable and developed a more positive image of the agency managing black bears. Participating stakeholders also tended to develop greater tolerance for other stakeholders with different interests as a result of becoming better educated about bear management. Similarly, wildlife professionals in Virginia developed more positive opinions about the role of stakeholders in the management process.

Recent experiences with stakeholders in controversial resource management issues demonstrate that resource managers must communicated openly and rapidly, using communication tools that stakeholders use. A process for addressing issues between landowners and hunters who use hounds in Virginia became a public relations nightmare for the state wildlife agency, because leaders of the hound hunting community successfully portrayed the effort as an attempt to ban or severely restrict hound hunting (even though the stated goal of the project was to ensure the future of hound hunting). Leaders of the hound hunting community used blogs, web forums, and other forms of electronic communication to effectively spread misinformation. Hound hunters believed the incorrect messages because a source they trusted (their peers) repeated the misinformation frequently and via methods that spread the misinformation quickly

I suggest that effective managers of wild trout in the future will need to do three things: (1) they should continue to focus on separating value choices from technical choices in the management process, (2) they should make technical choices more transparent, and (3) they should develop and use the communication tools their stakeholders use. The education that fisheries managers receive prepares them to make technical choices, but it gives them no more right to make values choices than their stakeholders. Fisheries managers can increase the transparency of technical choices by conducting data workshops, where professionals and stakeholders jointly review available data and develop agreement on the best available data for management of a fishery. Such a process is already in place in the marine fisheries arena (e.g., the Southeast Data Assessment and Review process used by the South Atlantic and Gulf Fisheries Management Councils). A recent survey by the Recreational Boating and Fishing Foundation found that anglers post about 500,000 messages a month about fishing to social media sites. Fisheries managers must update their communication strategies to include social media if they want to communicate quickly and effectively with future anglers.

CLIMATE CHANGE AND WILD TROUT: WHAT CAN OR SHOULD WE DO ABOUT IT?

Bruce Rieman and Dan Isaak
USDA Forest Service Rocky Mountain Research Station, Boise Aquatic Sciences Laboratory

There is now little question that earth's climate is changing and that human causes are fundamentally important. Periodic assessments of the International Panel on Climate Change (IPCC 2007) synthesize the most comprehensive evidence on the nature of global climate change, causes, scientific uncertainty, and the implications for human and natural systems. A rapidly expanding literature, instrumental record, and other work (e.g., http://cses.washington.edu/cig/fpt/cloutlook.shtml; http://www.fs.fed.us/ccrc/) document changes already occurring as well as observed and potential effects on natural resources.

Effects of climate change could be particularly profound for wild trout and aquatic ecosystems. Warming air temperatures and changing precipitation translate to increasing water temperatures, alteration of stream hydrology, and changes in the frequency, magnitude, and extent of extreme events such as floods, droughts, or wildfires. The biology of salmonids is largely dependent on temperature and flow and most have evolved in a dynamic environment defined by these processes. The upshot is that fundamental changes in climate could lead to fundamental changes in physiology, behavior, and growth of individuals, phenology, growth, dynamics and distribution of populations, persistence of species and structure of communities, and the functioning and services provided by whole ecosystems including fisheries for wild trout.

In relation to environmental variation over evolutionary and ecological time scales that shaped existing species of fishes and phenotypic diversity within those species (~10^6 - 10^3 yr, respectively), contemporary changes in climate may be relatively minor. Indeed many species such as the salmon, trout and char, that are now a focus of intensive management and recovery efforts, have recovered, diversified, and expanded in distributions following continental glaciation, volcanism, and the cataclysmic processes that shape mountain streams and rivers. The effects of contemporary climate change, in contrast to the degradation and loss of aquatic habitats associated with human development over the last two centuries, might represent a relatively minor loss of historical habitat capacity or productivity. The problem is that modern climate change is occurring especially quickly, at the end of an already warm period (IPCC 2007) indicating potential for conditions with no natural precedent and in the wake of already extensive habitat and aquatic community disruption. Many species and populations no longer have the networks and diversity of habitats and refugia, genetic diversity or evolutionary potential that allowed them to resist or rebound in the face of past environmental disturbance and change. In some cases, anticipated changes may outpace the capacity for adaptation and dispersal that does still exist.

Vulnerability of wild trout populations and fisheries to climate change will depend on a context defined by the characteristics of the species and populations, local environments, past habitat disruption, fragmentation and loss, and the nature of the change that occurs. As a result, that vulnerability will vary dramatically across populations, species, and landscapes that are a focus of land and natural resource management. In most cases capacity for conservation management is constrained by limited budgets, time, and other resources. It is also constrained by current understanding of the implications of climate change and management actions or alternatives that might effectively influence results. It will be important to prioritize limited resources and to guide management based on some understanding of the vulnerability of species, populations, and ecosystems of interest.

To consider management alternatives in the face of climate change, we synthesized information on climate change and salmonids, stream habitats, and the observed and anticipated effects of climate change in the intermountain west. This region may be particularly sensitive to climate change given the rate at which it is warming and because many of its ecosystems are constrained by water availability. We did not provide an exhaustive review of the climate-aquatic-fisheries literature-- several good ones already exist-- but rather developed an overview of important information as context for management that might begin to address the most relevant issues. Our report is organized around the following questions: *What is changing, what are the implications for wild fishes*, and *what can we do about it?*

In this paper we provide a brief overview of the key points to emerge from our synthesis and briefly explore recommendations for how we might *anticipate*, *adapt*, *prioritize*, *and collaborate* in management of wild trout in response to climate change. The full report (Rieman and Isaak, In Press) will be published this fall.

References

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate change 2007: the physical science basis. [Online]. Available: http://www.ipcc.ch/ [January 21, 2009].

Rieman, B.E. and Isaak, D.J. In Press. Climate change, aquatic ecosystems and fishes in the Intermountain West: Implications and alternatives for management. USDA Forest Service, Rocky Mountain Research Station, General Technical Report, GTR-000, Fort Collins, CO.

Wild Trout X – Closing Remarks

Steve Moore
Supervisory Fishery Biologist, Great Smoky Mountains National Park, Gatlinburg, Tennessee

History — An Opportunity to Share Information

It is always a treat to attend a Wild Trout Symposium – personally and professionally. I find it to be one of the best conferences I attend. The program and the variety of papers and posters presented on the many facts of wild and native trout management are always fascinating. The information exchanged and the professional and personal friendships that come from this meeting last a life time. Past and present conference organizers should be proud of the results of their efforts. I can vividly recall my first Wild Trout Conference in 1989. I was awed by the fact that I met and talked with many biologists whose work I had read and studied during my masters program and early career. Many of those folks became mentors as well as friends. Over the years, as I became more involved with the organization of these conferences, I gained even more respect for the biologist who came and presented the results of their work at these symposia and hope for the future of native and wild trout.

As I look at the history of this conference, there have been different themes. But all of them share common threads — the protection and preservation of native and wild trout. What type of regulations work best, there is no silver bullets, and how do we work with constituency groups to increase awareness and increase efforts to work together? Throughout this time frame biologists across the continent and worldwide have done much to understand and restore habitat where feasible and to focus on the ecology of native and wild trout. This work has increased awareness of and appreciation for native and wild trout and the fragile environments that sustain them. This symposium has added to our awareness of excellent science being conducted to add to our knowledge of wild trout and angler values. The sessions and papers were excellent and I applaud the organizers for their efforts.

Because of these factors there is a growing support for management strategies within agencies that favor the management and protection of native and wild trout populations and the restoration of habitat that will support them. Accompanying this trend is recognition of the inherent value of fishing for and observing wild trout by some anglers.

Observations — We are Always Under the Microscope

One observation during my career is that despite our best efforts to protect and preserve wild trout, we are under the microscope of the public and often that of politicians. Government employees (state or federal) are not trusted. Our actions, no matter how small or large, are scrutinized and criticized. **Why?** I am sure there are a variety of reasons. The Plenary Session provided an excellent overview of ways to gather information about angler and constituent expectations. But it also challenged us to clearly **communicate** the scientific information we have to constituency groups and the public. The presenters also challenged us to make every effort **to understand and manage angler expectations** or those of other user groups (i.e., human dimensions). Lastly, we must **make sure that stakeholders and professionals have a role in decision making**. This can be accomplished by making holding workshops to share data and agency mandate with all user groups and the media to ensure transparency. These steps will help develop **mutual trust and good working relationships.**

Make Connections with Environmental Factors that Society Values

In past conferences and this one, I have heard a lot about future threats to wild trout, range loss, habitat fragmentation, invasive species, climate change, and the majority of the population that does not care about wild trout. In fact, in this conference it was mentioned that only 2% of the population fishes. The question was . . . does the other 98% know or care? The challenge that was issued was to **make connections with environmental factors that society values**. Many in the public may not fish, but they equate trout with clean water. Society recognizes the need for a healthy environment; our challenge as resource stewards is to increase their awareness that clean air and clean water is necessary for all life on this planet as we know it. We as biologist cannot do this alone; we need to develop collaborate efforts with non-traditional partners to broadcast this message. Our message must be consistent and we must all have the same goal to protect the environment that supports diverse aquatic ecosystems.

Together We can Make a Difference

Historically, I know that I have left this meeting ready to help make a difference. Then I get home, and a week of phone calls and emails need my attention. Then there are agency issues to deal with and the wife, kids, and grandkids still need some of my attention too – and I need theirs! These priorities quickly move my commitment to the new task close to the bottom of the priority list. With luck, they may resurface. The last thing any of us need is something else to do. But, if we **wish** to make changes in the public arena we must realize that none of us alone can make a difference then this needs to become a priority — not only for each of us individually, but for our agency, perhaps AFS and partner groups like TU, FFF and others.

One potential option is to find an issue in your region that all agencies agree is a high priority. Then define how you can move forward. I'll use an example from the Smokies. In 2006 the state of Tennessee listed 12 streams in the Park as acid impaired. Given that the Park owns the top of the watershed, the only source of pollution is acidic deposition. Tennessee Department of Environment and Conservation completed a TMDL (Total Maximum Daily Load) for the impairment and we are in the process of developing an implementation plan for the TMDL. As you know, air pollution is not confined to Tennessee. This issue affects the Southeast and most likely the east coast. There are many agencies and universities working on this acid deposition issue and its effects on aquatic resources. There is and has been some information exchange but I was amazed to learn how much work is really going on in this area. But, I was dismayed that we were not communicating more and that many agency leaders and constituent groups did not know about the potential resource damage, like declining pH in many streams and the loss of brook trout in headwater streams in Great Smoky Mountains National Park. If the land managing agencies in the Southeast were to make this a priority, share data and have a standard language then political and public awareness could be raised.

I know that this issue or any other that is chosen will NOT be this simple and it will be a long hard journey. **But, if we do not try who will?** We have heard a lot about climate change and the potential for future loss of wild and native trout resources. That is a long-term problem that will play out over decades. Do we need to be thinking about this issue and taking proactive steps to collect good monitoring data? YES! But, we need to keep things in perspective. There are numerous immediate threats and issues that we can fix in the near term. My guess is that you know what those are and hope to make progress on those as soon as possible. I encourage you to have short- and long-term goals.

Last, for Wild Trout XI, I think it would be excellent if someone would take a look back and summarize the progress that has been made in the past four decades. I would also hope that many of you will return and report on the progress that you have made on increasing awareness of the value of native and wild trout resources and the diverse ecosystems that support them. Until then, have a safe trip home and best wishes for great work protecting wild trout and their habitat.

Session 1: Climate Change and Wild Trout

VULNERABILITY OF NATIVE TROUT HABITAT UNDER CLIMATE CHANGE: CASE STUDIES FROM THE NORTHERN ROCKY MOUNTAINS

Stephen T. Gray[1] and Gregory T. Pederson[2]

[1]Water Resources Data System and Wyoming State Climate Office, University of Wyoming, Laramie, WY, sgray8@uwyo.edu

[2]US Geological Survey, Northern Rocky Mountain Science Center, Bozeman, MT

ABSTRACT — The Northern Rocky Mountains (NRM) act as a critical headwaters for a large swath of North America. In turn, the NRM serves as a local stronghold for many native trout species, while also providing runoff that contributes to salmonid habitats far downstream. Using examples from a series of observational and modelling studies, we document recent trends in NRM hydroclimate, and investigate how these trends are linked to temperature change and potential impacts on native trout. NRM temperature records show modest (2.7 to 4.2°C) increases in minimum and maximum temperatures over the past 20-30 years, but this has resulted in a significant increase in days > 0°C and hot (e.g., >32°C) summer days. Likewise, observed temperature changes have coincided with significant shifts in the timing of snowmelt, increases in snow-free days, and altered streamflow timing, with impacts at higher elevations likely amplified by surface-albedo feedbacks. Modelling studies suggest that additional temperature increases on the order of just 1-2°C could lead to extremely low late-summer flows, while also greatly exacerbating the impacts of drought on regional streams. At the same time, simulations show that increases in winter precipitation within the bounds of most regional climate predictions would not offset the effects of warmer temperatures on these systems. Overall, these studies point to a range of future scenarios whereby rising temperatures could lead to significant late-summer drought and changes to the seasonal hydrograph, as well as major changes in regional water availability.

AUGUST STREAM DISCHARGE TRENDS PORTEND IMPACTS OF CLIMATE CHANGE IN THE NORTHERN ROCKIES

Jason C. Leppi[1*], Thomas H. DeLuca[2], Solomon Harrar[3], Steve W. Running[1]

[1]Numerical Terradynamic Simulation Group (NTSG), College of Forestry and Conservation, University of Montana, 32 Campus Drive, Missoula, MT, 59812, USA; [2]School of Environment Natural Resources, and Geography, 2nd Floor Environment Centre Wales, Bangor University Bangor, Gwynedd LL57 2UW,United Kingdom;[3]Department of Mathematical Sciences, University of Montana, Mathematics Building, Missoula, MT, 59812, USA.

*Corresponding author. Tel 406 579 0296; E-mail address: Jason.leppi@ntsg.umt.edu (J.C. Leppi).

ABSTRACT — In the snowmelt dominated hydrology of arid western U.S. landscapes, late summer low streamflow is the most vulnerable period for aquatic ecosystem habitats and trout populations. This study analyzes mean August discharge at 153 streams throughout the Northern Rockies (NR) for changes in discharge from 1950 to 2008. The purpose of this study was to determine if (1) mean August stream discharge values have decreased over the last half century; (2) low discharge values are occurring more frequently; and (3) climatic variables are influencing August discharge trends. Here we use a strict selection process to characterize gauging stations based on amount of anthropogenic impact to tease out heavily impacted rivers and understand the relationship between climatic variables and discharge trends. Using historic U. S. Geologic Survey (USGS) discharge data, we analyzed data for trends of 40 to 59 years. Combining of these records along with aerial photos and water rights records, we selected gauging stations based on the length and continuity of discharge records and categorized each based on the amount of diversion. Our analyses indicate that non-regulated watersheds are experiencing substantial declines in stream discharge and we have found that 89% of all non-regulated stations exhibit a declining slope. Additionally, our results indicate a significant ($\alpha \leq 0.10$) decline in discharge from 1951 to 2008 for the NR. Correlations at our pristine sites show a negative relationship between air temperatures and discharge, and these results coupled with increasing air temperature pose serious threats for aquatic ecosystems in NR.

Key words: Discharge, climate change, Northern Rockies, stream flow

INTRODUCTION

Water in the interior western United States is a vital resource with seasonally limited supply and an increasing demand; however, there is currently limited knowledge regarding summer discharge trends in the Northern Rockies (NR). Changes in the annual water balance that decrease the amount of water available to NR communities and ecosystems will negatively impact the region. Many communities depend on stream discharge to fill local reservoirs for drinking water, and without adequate discharge they will be forced to rely on groundwater and other sources. The NR serve as the headwaters of two major United States river basins, the Columbia and Missouri river basins, and changes in stream discharge in the NR will be felt throughout both basins. Aquatic species require adequate discharge and coldwater temperatures for survival. Local economies also depend on stream discharge for growing crops, raising cattle, and tourism income through guided fishing and rafting trips. Over the last few years the NR has experienced unprecedented hot summers and dry water years, which has escalated concerns over the survival of sensitive aquatic species and the potential impacts to ecosystem services and livelihood for the region.

The NR climate varies from semi-arid continental to highland depending on elevation, topography, and geographic location (Critchfield 1983). Most areas have cold winters and warm summer months with an average precipitation of 48.3 cm in which ~ 62% is received as snow (Serreze et al. 1999; Daly et al. 2008). Rivers and watersheds are snowmelt dominated, in that seasonal snowmelt during the spring and summer months are a critical part of each watershed. In snowmelt dominated areas many aquatic plants and animals depend on this seasonal pattern of discharge for reproduction and survival (Ficke et al. 2007). Due to these characteristics, the NR are particularly sensitive to changes in air temperature, and the estimated trends pose serious concern for the region. Hansen et al. (2006) estimated that annual air temperatures from 1950 to1998 have increased between + 0.5 and 1.0 Cº. Using data from weather stations in western Montana, Pederson et al. (2010) estimated that western Montana has experienced a rise in annual average temperature of + 1.33 Cº from 1900 to 2000 and even greater increases in minimum and maximum winter temperatures, which is similar to previous observations for the entire western U.S. (Bonfils et al. 2008). Evidence of the sensitivity to increasing air temperatures has been documented in numerous studies across the region including (1) decreases in snow water equivalent measurements (Mote 2003; 2005); (2) increases in the amount of winter precipitation as rain (Knowles et al. 2006); (3) earlier center of timing of snowmelt (Stewart et al. 2005; Moore et al. 2007); and (4) an increase in evaporation (Golubev et al. 2001; Brutsaert 2006). These documented changes in the water balance and the characteristics of NR watersheds that make August stream discharge an important target, because it provides an integrated view of the cumulative impacts of air temperature increases.

In the NR stream water temperatures are typically warmest when air temperatures are highest and stream discharges are lowest. Unfortunately, there is limited availability of historical water temperature data; therefore, the following research was performed to evaluate fluctuations in August stream discharge as a proxy for changes in stream temperature. Changes in discharge are correlated with water temperature due to the fact that as the volume of water decreases, the capacity of the river to buffer against temperature changes also decreases from reduced heat capacity associated with the decreased volume. Several previous studies have evaluated historical stream discharge trends throughout the U.S., but only a limited effort has been made in the NR. While annual flows are important measurements to help understand changes occurring within watersheds, they do not indicate when changes are occurring. Annual discharge assessments lump monthly discharge measurements into a 12- month package, which masks seasonal changes and does not provide a focused evaluation of the impacts on aquatic ecosystems.

Currently, only seven rivers in the NR within the U.S. have been analyzed for long-term trends during August (Rood et al. 2008). The work reported here will build upon previous research by analyzing a larger database of streams throughout the NR, using a more stringent selection process, and by analyzing more streams and a larger area, it allows us to better characterize summer discharge trends on a regional and local level. The purpose of the work reported was to determine if (1) mean August stream discharge values in the NR are declining; (2) low August stream flow values are occurring more frequently with time; and (3) there is a correlation between climatic variables and August discharge at pristine sites

Methods

The NR study area encompasses Montana, Idaho, and Wyoming. Data were available from a total of 2,895 USGS gaging stations, but only those stations that met specific criteria were selected for inclusion in our study. This strict selection process differs from previous studies, because we only accepted stations that had a minimum of 40 years of semi-consecutive (< 2 years of consecutive missing data) data were chosen. We chose the past 59 years to analyze to capture the impacts of a ~1ºC air temperature increase that occurred during this period in the Northern Rockies (Hansen et al. 2006; Pederson et al. 2010). Sites were then grouped into three categories based on the amount of anthropogenic impact by analyzing dam records, surface water diversion records, and land use information using satellite imagery (< 10 m resolution), water rights records and GIS. Sites were grouped as pristine, unregulated, and regulated, and the major difference among these sites is the degree of anthropogenic impact. We thus ended up with a subset of 15 pristine sites that is unique from the Hydro-Climatic Data Network (HCDN) sites used in previous research (Slack and Landwehr 1999), because by using water diversion records we were able to more accurately select relatively unaltered streams. The HCDN sites were not used, because upon inspection many of the HCDN sites contained small dams and diversions upstream of the gauging station as noted in previous research (Moore et al. 2007). An overview of our site characteristics can be seen in Table 1. Historical discharge records were collected from the USGS national water information system web interface (http://waterdata.usgs.gov/usa/nwis/sw). The web interface allows the user to access historical stream information from 23,777 stations across the U. S. December to July mean temperature and precipitation model output records for the watersheds upstream of the 15 pristine gauging stations were collected from WestMap's Climate Analysis and Mapping Toolbox, which uses data computed by the PRISM Model (http://www.cefa.dri.edu/Westmap/). Due to the remote locations of many of the sites and limited number of weather stations in the region, PRISM model output provides the best available estimate for these parameters (Daly et al. 2008).

Data Analysis

We employed a trend-free, pre-whitening (TFPW) procedure (Yue et al. 2002b) to account for serial correlation and the Mann-Kendall test was used to examine the data for a trend (Mann 1945; Kendall 1975; Yue et al. 2002b). An alpha level of 0.10 was used to assess statistically significant trends with the Mann-Kendall test. To normalize the discharge data, the slope of each site was divided by the mean discharge of the same gauging station in order to compare change in discharge between streams. Once a local statistical significance was determined, we used a bootstrap re-sampling technique developed by Douglas et al. (2000) to examine the field significance (Statistical significance of a region) of discharge trends within our site groupings and across the NR. To examine the effect of cross correlation, i.e., falsely rejecting the null hypothesis, we employed two types of bootstrap re-sampling methods on TFPW data over two time periods: 1957-1996 and 1951-2008 (Douglas et al. 2000; Yue et al. 2002b). We employed the block-bootstrap method, which takes into account spatial correlation, to determine field significance (Douglas et al. 2000). To analyze low discharge frequency, the median and the lower and upper limit of the interquartile range were calculated for all gauging sites. Once the upper and lower limits of the interquartile range were found, low discharge frequency was focused on to determined the frequency over time that discharge values were less than the 25th percentile. A five-year moving average was then superimposed over the frequency bar plot to visually detect trends.

To examine the relationship between temperature and precipitation to discharge at the pristine watersheds, Spearman and Kendall correlation analyses were performed. To better determine how climatic variables influence August discharge, we developed a linear mixed effect model to model discharge. This model was created in R statistical program using the NLME library (Pinheiro and Bates 2000) and included December-July precipitation and air temperature as well as drainage area as independent variables. We examined simple and complex models and through an Akaike Information Criterion (AIC) analysis determined a best fit model. We also examined the variables in the model for multicollinearity and did not find a significant relationship.

Table 1. Comparison of gauging site groupings in the Northern Rockies. The analysis indicates that the largest decreases in discharge are occurring in the pristine and unregulated sites over the 1950-2008 period.

Site Classification	Number of Sites	Number of Significant Trends		Change Per Record	
		Decreasing	Increasing	Mean	Median
Pristine	16	8	0	-22.87%	-22.15%
Unregulated	49	13	2	-18.73%	-21.58%
Regulated	88	14	11	-9.44%	-11.84%

Results

Non-regulated Sites

Regional analysis of discharge trends shows a strong decreasing pattern across the NR. By examining the distribution of the normalized discharge (slope divided by average discharge), it is possible to determine where the distribution is centered. If the distribution was centered at a value greater than zero, discharge trends would be expected to be increasing with time. However, the distribution of all the sites examined (n = 65) is centered at -21% change, and further examination reveals that 89% of all sites had negative slopes. Our bootstrap analysis revealed that field significance (statistical significance for a region with more than one gauging site) for each grouping of gauging sites and the entire NR had significant decreasing trends over the full time period (1950-2008). Field significance was not observed when more sites were examined over a shorter time period (1957-1996). Widespread decreasing trends provide strong evidence that discharge is decreasing throughout the NR (Figure 1) and provides much needed information for critical August stream flows in this region.

Further examination of changes over different time periods provides supporting evidence of decreasing trends. A similar pattern in the discharge slope was observed for the 58- and 38-year trends of sites suggesting that with time discharge change is uniform with a slight decrease toward the end of the 20st century and beginning of the 21st century. When other summer months are examined, such as July and September, we found similar patterns to August trends. Both of these patterns suggest that in addition to decreasing August trends, there is a general pattern of decreasing summer discharge (July-September) at most sites.

Pristine Sites

From the non-regulated sites, we selected a subset of sites deemed pristine (gauging stations without reservoirs or surface water diversions upstream). All discharge records analyzed have a decreasing slope and the median change over the record length is -23%. Eight of 15 sites have significant ($\alpha \leq 0.10$) decreasing trends, and no sites were observed to have an increasing trend. Additionally, bootstrap results reveal that the region is significantly ($\alpha \leq 0.10$) decreasing. December through July temperature records show strong statistical evidence for increasing temperature trends in 14 of the 15 watersheds analyzed. December-July precipitation for most sites is decreasing, but only one of 15 sites had a significant decrease in precipitation during the period of record. There is a negative correlation

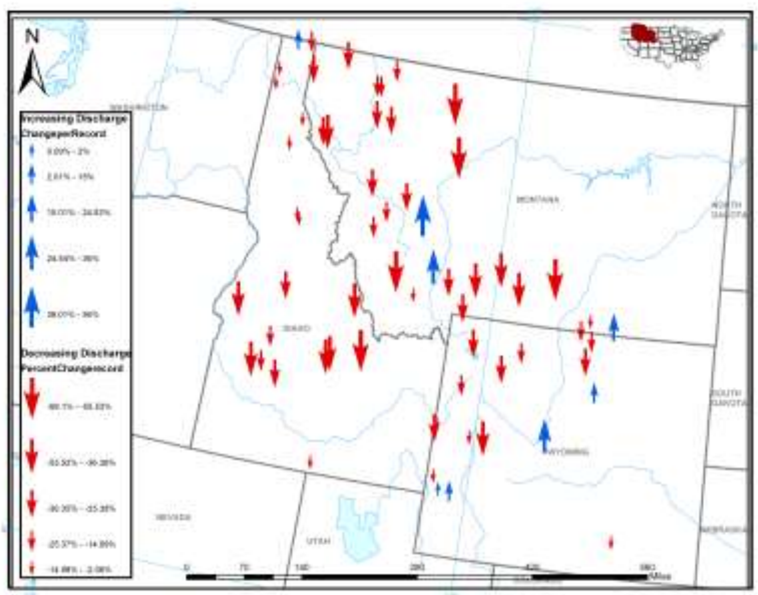


Figure 1. Amount and type of normalized discharge change per record across the Northern Rockies. The downward pointing red arrows signify a decreasing slope and the upward pointing blue arrows signify an increasing slope. The larger the arrow the larger the discharge change at each gauging station. This figure shows a decreasing trend across the study area with very few positive slopes.

between air temperature and discharge, and a positive correlation between precipitation and discharge.

The linear mixed effects best model created for the entire NR and each pristine gauging station indicates that precipitation, temperature, and drainage are statistically significant predictors of August discharge and that by incorporating random effects and auto-correlation and accounting for heteroscedasticity August discharge was modeled with greater precision than models that use simple linear regression. We also found that within our model air temperature had the greatest effect on August discharge.

Discussion: Implications for the Northern Rockies

Analyses of long-term stream discharge records across the NR indicate that August discharge has generally declined over the last half century, the frequency of low discharge values has been greater in the last eight years than previous decades, and changes in air temperature cause or at least influence changes in August discharge. While discharge values are not significantly decreasing in every location, our results indicate that there is sufficient evidence to suggest that discharge is declining across the region. Many factors will influence the identification of a significant trend and it may be just as important to look at the change over time and the practical significance. Our intent in studying discharge trends was to improve knowledge regarding trends in stream discharge for the NR which could be applied to issues in conservation and land management. By separating streams based on the type and amount of anthropogenic control or modification, we feel that we greatly improved our capacity to identify areas in the NR where climate change is directly influencing stream discharge. In contrast, those areas that are heavily impacted by human activities can be evaluated for the occurrence of trends, but there is little chance of identifying the cause of these changes. Observation of changing trends in human impacted systems is of great interest; however, teasing out the causality of discharge trends in unregulated and regulated sites is a complex task that was not within the scope of this study. Detection of trends change depending on the type of analysis, length of data, and intensity or type of change (Kundzewicz and Robson 2004; Radziejewski and Kundzewicz 2004). The significance of trends also changes depending on how the analyses are conducted (Daniel 1978; Yue et al. 2002a).

While statistical significance is a common means of differentiating long-term trends from patterns that occur by chance, information from trends that are not statistically significant can also be useful and sometimes statistically significant trends are not useful information (Daniel 1978; Yue et al. 2002a). For example in the analysis of discharge trends in this study, we analyzed the normalized change over time (normalized slope) as well as the significance of the trend and found that a number of stations displayed large decreases in slope, but these trends were not statistically significant (at α level 0.10). It has been stated in previous research (Daniel 1978; Yue et al. 2002a) that this is a common pattern in hydrological analysis, and we found several stations that had large changes in discharge, but were not statistically significant. Locations that have large changes over time are important information for scientists and mangers in the NR, and these sites should not be disregarded as unimportant. By performing a detailed screening of gauging sites, based on amount of anthropogenic impact, we were able to tease out additional impacts from humans that influence stream discharge levels. Data from all pristine sites have decreasing slopes and eight of 15 sites have statistical evidence for a trend ($\alpha \leq 0.10$). The data from these stations show statistically that a decline is occurring from influences beyond local human change in surface water diversions, dam regulation, and land use change. A spatial pattern for discharge declines is not evident for non-regulated sites, but it appears that the largest declines are occurring on relatively small (~100-200 mi^2) watersheds. This may be due to the fact that rivers with large watersheds are more influenced by groundwater or those larger watersheds are more buffered to changes in hydrologic inputs due to the sheer number of sub-drainages.

Trends from our regulated sites are additionally important information on the overall status of streams in the area, but caution should be used in interpreting these results because they are extremely manipulated by energy production at dams, irrigation demand, irrigation recharge, and water rights downstream. From our regulated sites (n = 88), or sites regulated by dams, we found 67% of the regulated sites had a negative slope with a median change over the record of -11.8%. Our bootstrap analysis revealed that field significance for regulated gauging sites and the entire Northern Rockies had significant decreasing trends over the full time period (1950-2008). These results appear to show that August discharge at regulated sites, at certain locations, are declining less across the region. However, these results cannot be interpreted directly because the locations of regulated gauging sites are not distributed randomly across the terrain. Regulated sites tend to be located on rivers with larger drainage areas and may be more controlled by groundwater or buffered by the sheer number of subwatersheds feeding into the basin causing August discharge to remain relatively constant.

Our analysis builds upon past research (Hamlet et al. 1999; Lins and Slack 1999; Rood et al. 2005; 2008; Luce and Holden 2009) by examining a larger area, more locations and identifying sites that have additional impacts that may influence discharge trends. We specifically focused on August discharge trends, because changes in August discharge have greater potential ecological consequences than other months of the year. In comparison to past research, our results show that decreases in August discharge trends are occurring throughout the NR and are not limited to areas east of the continental divide. Additionally, our results of increasing low flow values (< 25th percentile) supports work by Luce and Holden (2008) who state dry years are becoming substantially drier in the region. These results are disconcerting for communities and aquatic ecosystems throughout the NR.

Impacts on Fisheries

Discharge in rivers is a critically important water quality parameter that helps maintain cool water temperatures, adequate dissolved oxygen levels, and suitable habitat for aquatic biota. When discharge drops, streams are more susceptible to elevated air temperatures than at times of high flow. Adequate discharge is most important during summer months when air temperatures are above 32ºC, thus posing the risk of heating rivers above the suitable temperature for various aquatic species. Current and future estimates in average annual air temperature from historical temperatures are expected to increase in the NR making many streams candidates for increases in stream temperatures (Bales et al. 2006; Trenberth et al. 2007; Barnett et al. 2005; 2008). Species most susceptible to elevated water temperatures in the NR are native fish from the salmonoid family such as bull trout *Salvelinus confluentus*, cutthroat trout *Oncorhynchus clarkii*, rainbow trout *Oncorhynchus mykiss*, arctic grayling *Thymallus arcticus* and various salmon species: Chinook *Oncorhynchus tshawytscha*, coho *Oncorhynchus kisutch,)* and sockeye *Oncorhynchus nerka* salmon. All salmonoids have an optimum temperature for growth with a minimum and maximum temperature range where conditions for growth are adequate, but not optimal. Typically, growth curves for salmonids increase moderately to an optimal water temperature and then decrease rapidly, thus making fish more susceptible to harm from increases in temperature. As native species migrate or decrease in abundance, it is predicted that many nonnative species, with higher fundamental thermal niches, will move into areas previously or sparsely inhabited by native species (Carveth et al. 2006). Our results provide evidence of decreasing discharge patterns across the region and in combination with expected warmer air temperatures many coldwater species may be negatively impacted from increases in stream temperatures.

CONCLUSIONS

Our analyses indicate that non-regulated watersheds throughout the NR are experiencing significant declines in stream discharge over the last half century. Our research confirms that there is a significant decreasing trend in August discharge occurring across the region, and low discharge frequency has been greater in the last 8 to 10 years than previous decades. Our research also confirms that changes in air temperature influenced changes in August discharge trends for the NR over the last half century. These results support previous research (Rood et al. 2005; 2008) which has indicated that decreasing discharge trends are occurring across the region and the driest of years are becoming drier with time (Luce and Holden 2009). However, by examining a greater area and number of stations, our research indicates that decreasing August discharge trends are not limited to stations east of the Continental Divide. Through new selection techniques we were also able to analyze stations that had minimal anthropogenic impact on watersheds and by doing this we greatly improved the hope of identifying areas in the NR where climate change is causing stream discharge to decrease over time.

Our results have serious implications for aquatic ecosystems and communities in the NR. Decreasing August discharge in snowmelt-dominated watersheds has the potential to severely impact many aquatic flora and fauna in the region. Less water in rivers during summer months will increase competition for water resources in the area and make management of resources difficult. While this research confirms a declining trend across the region, more research on the causality of trends is needed to explain the extent to which climate change is affecting unregulated and regulated watersheds in the region. With air temperatures projected to continue increasing into the future, it is of great concern to understand if August discharge trends will follow observed trends or be exacerbated by elevated air temperatures, thus making water resources even more limited.

Literature Cited

Bales, R.C., N.P. Molotch, T.H. Painter, M.D. Dettinger, R. Rice & J. Dozier. 2006. Mountain hydrology of the western United States. Water Resources Research 42: 13.

Barnett, T.P., J.C. Adam & D.P. Lettenmaier. 2005. Potential impacts of a warming climate on water availability in snow-dominated regions. Nature 438: 303-309.

Barnett, T.P., D.W. Pierce, H.G. Hidalgo, C. Bonfils, B.D. Santer, T. Das, G. Bala, A.W. Wood, T. Nozawa, A.A. Mirin, D.R. Cayan & M.D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319: 1080-1083.

Bonfils, C., B.D. Santer, D.W. Pierce, H.G. Hidalgo, G. Bala, T. Das, T.P. Barnett, D.R. Cayan, C. Doutriaux, A.W. Wood, A. Mirin & T. Nozawa. 2008. Detection and Attribution of Temperature Changes in the Mountainous Western United States. Journal of Climate 21: 6404-6424.

Brutsaert, W. 2006. Indications of increasing land surface evaporation during the second half of the 20th century. Geophysical Research Letters 33: 4.

Carveth, C.J., A.M. Widmer & S.A. Bonar. 2006. Comparison of upper thermal tolerances of native and nonnative fish species in Arizona. Transactions of the American Fisheries Society 135: 1433-1440.

Critchfield, H.1983. General Climatology. Prentice-Hall. 145-161

Daly, C., M. Halbleib, J.I. Smith, W.P. Gibson, M.K. Doggett, G.H. Taylor, J. Curtis & P.P. Pasteris. 2008. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology 28: 2031-2064.

Daniel, W.W. 1978. Applied Nonparametric Statistics, Boston.

Douglas, E.M., R.M. Vogel & C.N. Kroll. 2000. Trends in floods and low flows in the United States: impact of spatial correlation. Journal of Hydrology 240: 90-105.

Durbin, J. & G.S. Watson. 1950. Testing for serial correlation in least squares regression.1. Biometrika 37: 409-428.

Ficke, A.D., C.A. Myrick & L.J. Hansen. 2007. Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries 17: 581-613.

George, S.S. 2007. Streamflow in the Winnipeg River basin, Canada: Trends, extremes and climate linkages. Journal of Hydrology 332: 396-411.

Golubev, V.S., J.H. Lawrimore, P.Y. Groisman, N.A. Speranskaya, S.A. Zhuravin, M.J. Menne, T.C. Peterson & R.W. Malone. 2001. Evaporation changes over the contiguous United States and the former USSR: A reassessment. Geophysical Research Letters 28: 2665-2668.

Hamlet, A.F. & D.P. Lettenmaier. 1999. Effects of climate change on hydrology and water resources in the Columbia River basin. Journal of the American Water Resources Association 35: 1597-1623.

Hansen, J., M. Sato, R. Ruedy, K. Lo, D.W. Lea & M. Medina-Elizade. 2006. Global temperature change. Proceedings of the National Academy of Sciences of the United States of America 103: 14288-14293.

Hirsch, R.M., J.R. Slack & R.A. Smith. 1982. TECHNIQUES OF TREND ANALYSIS FOR MONTHLY WATER-QUALITY DATA. Water Resources Research 18: 107-121.

Knowles, N., M.D. Dettinger & D.R. Cayan. 2006. Trends in snowfall versus rainfall in the Western United States. Journal of Climate 19: 4545-4559.

Kendall, M.G. 1975. Rank Correlation Methods. Griffin, London.

Lins, H.F. & J.R. Slack. 1999. Streamflow trends in the United States. Geophysical Research Letters 26: 227-230.

Luce, C.H. & Z.A. Holden. 2009. Declining annual streamflow distributions in the Pacific Northwest

United States, 1948-2006. Geophysical Research Letters 36.

Mann, H.B. 1945. Non-Parametric tests against trend. Econometrica 13: 245–259.

Moore, J.N., J.T. Harper & M.C. Greenwood. 2007. Significance of trends toward earlier snowmelt runoff, Columbia and Missouri Basin headwaters, western United States. Geophysical Research Letters 34.

Mote, P.W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30: 4.

Mote, P.W., A.F. Hamlet, M.P. Clark & D.P. Lettenmaier. 2005. Declining mountain snowpack in western north America. Bulletin of the American Meteorological Society 86: 39-+.

Novotny, E.V. & H.G. Stefan. 2007. Stream flow in Minnesota: Indicator of climate change. Journal of Hydrology 334: 319-333.

Pederson, G.T., L.J. Graumlich, D.B. Fagre, T. Kipfer & C.C. Muhlfeld. 2010. A century of climate and ecosystem change in Western Montana: what do temperature trends portend? Climatic Change 98: 133-154.

Pinheiro, J.C, & D.M. Bates. 2000. Mixed-Effects Models in S and S-PLUS, Springer.

Radziejewski, M. & Z.W. Kundzewicz. 2004. Detectability of changes in hydrological records (vol 49, pg 39, 2004). Hydrological Sciences Journal-Journal Des Sciences Hydrologiques 49: 351-352.

Rood, S.B., J. Pan, K.M. Gill, C.G. Franks, G.M. Samuelson & A. Shepherd. 2008. Declining summer flows of Rocky Mountain rivers: Changing seasonal hydrology and probable impacts on floodplain forests. Journal of Hydrology 349: 397-410.

Rood, S.B., G.M. Samuelson, J.K. Weber & K.A. Wywrot. 2005. Twentieth-century decline in streamflows from the hydrographic apex of North America. Journal of Hydrology 306: 215-233.

Serreze, M.C., M.P. Clark, R.L. Armstrong, D.A. McGinnis & R.S. Pulwarty. 1999. Characteristics of the western United States snowpack from snowpack telemetry (snotel) data. Water Resources Research 35: 2145-2160.

Slack, J.R., & J.M. Landwehr.1992. Hydro-climatic data network: a U.S. Geological Survey streamflow data set for the United States for the study of climate variation. U.S. Geological Survey open file rept 1874-1988.

Stewart, I.T., D.R. Cayan & M.D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18: 1136-1155.

Trenberth, KE, P.D. Jones, P. Ambenje, R. Bojariu, D. Easterling, A. Klein Tank, D. Parker, F. Rahimzadeh, J.A. Renwick, M. Rusticucci, B. Soden, & P. Zhai, 2007. Observations: Surface and Atmospheric Climate Change. In: Climate Change 2007: The Physical ScienceBasis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Yue, S., P. Pilon & G. Cavadias. 2002a. Power of the Mann-Kendall and Spearman's rho tests for detecting monotonic trends in hydrological series. Journal of Hydrology 259: 254-271.

Yue, S., P. Pilon, B. Phinney & G. Cavadias. 2002b. The influence of autocorrelation on the ability to detect trend in hydrological series. Hydrological Processes 16: 1807-1829.

Yue, S.,P. Pilon, B. Phinney & G. Cavadias. 2002c. Patterns of Trend in Canadian Streamflow. Proceedings if the 58th Annual Eastern Snow Conference, GCU, May 14-17th, Ottawa, Ontario, Canada (in press).

FISHERIES MANAGEMENT DURING UNCERTAIN TIMES: DEVELOPING A DIVERSE CONSERVATION PORTFOLIO FOR THE LONG-TERM PERSISTENCE OF NATIVE TROUT

Jack E. Williams[1] and Amy L. Haak[2]

[1]Trout Unlimited, 329 Crater Lake Avenue, Medford, OR 97504

[2]Trout Unlimited, 910 W. Main Street, Suite 342, Boise, ID 83702

ABSTRACT — Management strategies that maximize biological diversity and promote varied approaches to population protection are more likely to succeed during a future where global warming and invasive species drive rapid environmental change and increasing uncertainty. We describe how the concept of a diverse management portfolio can be applied to native trout conservation by providing for Representation (protecting and restoring diversity), Resilience (having sufficiently large populations and intact habitats to survive environmental change), and Redundancy (saving enough different populations so that some can be lost without jeopardizing the species). Saving diversity for native trout requires the conservation of genetically pure populations, the protection and restoration of life history diversity, and the protection of populations across the historic range. Protecting larger, stronghold populations is important because such populations will have a better chance of surviving future disturbances, including those associated with climate change. The long-term persistence of populations is likely to require maintenance of larger population sizes and habitat patches than currently exist for many populations. Redundancy among these elements is important given that many populations are small, occupy reduced habitat in fragmented stream systems, and therefore are vulnerable to extirpation. Application of the concept is further described in case studies of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* and Rio Grande cutthroat trout *O. c. virginalis*, two subspecies that illustrate many of the management challenges common to western native trout.

"Interestingly, the parallels between issues of diversification of securities and issues of natural diversity are generally not acknowledged. This is surprising, because the correlation between diversity and stability in ecology has long been discussed…"--Frank Figge (2004:828)

INTRODUCTION

One of the basic tenants of conservation biology is that biological diversity provides stability to ecosystems (Primack 2002). Biologically rich communities are better able to withstand disturbance and swings in environmental conditions that would destabilize communities dominated by few species or populations. The ability of diverse natural systems to maintain their function and productivity in the face of rapid environmental change has been termed the "portfolio effect," a concept analogous to the desire among financial managers to maintain a diverse economic portfolio as a hedge on uncertain futures (Figge 2004). For natural resources, increasing the biological diversity within a management portfolio decreases the risk of failure of the management approach (Figure 1).

Among the factors that have made western trout so successful are their diverse life histories and wide range of occupied habitats. Migratory life histories facilitate access to diverse habitats. Such migratory lifestyles allow fish to avoid streams degraded by disturbances and subsequently reoccupy streams once habitats are restored. Many native trout historically occupied lakes of various sizes, elevations, and water qualities. Even stream habitats occupied by individual subspecies varied greatly. Bonneville cutthroat trout *Oncorhynchus clarkii utah*, for example, ranged from small low-elevation streams in the West Desert along the Nevada-Utah border to streams draining the High Uinta Mountains in north-

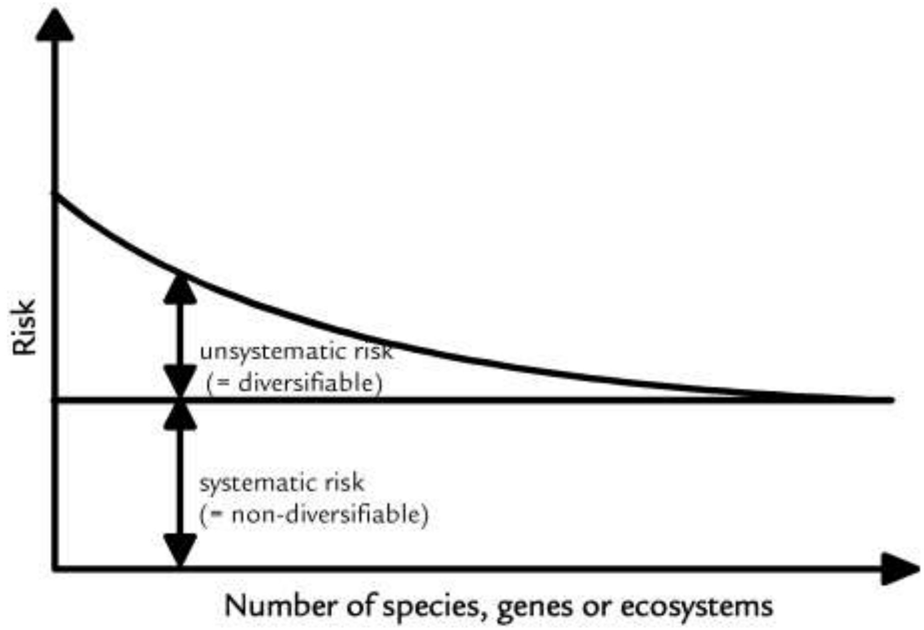


Figure 1. Typical risks to management portfolio as a function of increasing number of biodiversity elements. Note that there is some point, shown here as non-diversifiable risk, where the risk of catastrophic failure cannot be eliminated despite increasing diversity (figure from Figge 2004).

ern Utah. Various authors have described the importance of preserving life history diversity and habitat connectivity in our efforts to sustain native trout (Rieman and Dunham 2000; Colyer et al. 2005; Neville et al. 2009).

The importance of a diverse conservation portfolio to maintaining long-term productivity in fisheries was illustrated recently for Alaska's Bristol Bay sockeye salmon *Oncorhynchus nerka* fishery. Schindler et al. (2010) showed how high levels of population and life history diversity enabled the fishery to maintain a high productivity despite swings in environmental condition. Under one set of environmental conditions, a certain group of sockeye salmon populations would prosper, whereas under another climate regime, a different set of populations would prosper (Schindler et al. 2010).

Rapid global warming is likely to have significant negative impacts on most native salmonids (Haak et al. 2010b), especially those that have lost substantial amounts of their historic biological diversity. Not only will rising air temperatures increase the temperatures of lakes and streams, but also the frequency and intensity of disturbances such as flooding, drought, and wildfire (Poff 2002; Williams et al. 2009; Haak et al. 2010b). Already in many parts of the western U.S., stream runoff is peaking earlier in the year and flows are declining as compared to conditions from the early 20th Century (Luce and Holden 2009; Clark 2010). Small headwater trout populations will be particularly vulnerable to increased disturbance regimes because of their small population sizes and isolation (Brown et al. 2001; Williams et al. 2009). Furthermore, Rahel and Olden (2008) describe how climate change is likely to provide for new pathways of introduction for aquatic invasive species and increase the incidence of whirling disease and other pathogens.

In this paper, we describe a management approach designed to achieve a diverse management portfolio for native trout. This 3-R approach is representation, resilience, and redundancy. We show how this approach can be applied to Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* and Rio Grande cutthroat trout *O. c. virginalis*. These two subspecies represent much of the range of geology, topography, hydrology, and population structure that characterize cutthroat trout. We argue that strategic implementation of such an approach will increase the likelihood that native trout populations will persist in the future, even if this future is characterized by rapid environmental change and uncertainty.

Diversifying the Management Portfolio

For western native trout, a diverse management portfolio includes at least some proportion of the life history, habitat, genetic, and population diversity that has allowed these fishes to succeed and persist over time despite disturbances and changes to their environment. Diversity is key and reliance on one life history form or habitat type should be avoided. The concept can be illustrated in the use of in-stream barriers to isolate native trout from downstream nonnative species. On the one hand, isolation of populations may provide the best available protection from hybridization with nonnative trout, but at the same time, isolation of small populations will increase their vulnerability to extirpation from disturbance (Fausch et al. 2006). Therefore, a diverse management portfolio would include some populations isolated above barriers but others in larger, more interconnected stream systems.

Diversifying the management portfolio can be described via the 3-R strategy of **Representation** (protecting and restoring diversity), **Resilience** (having sufficiently large populations and intact habitats to survive environmental change), and **Redundancy** (saving enough different populations so that some can be lost without jeopardizing the species) (Shaffer and Stein 2000). The U.S. Fish and Wildlife Service adopted these principles in developing recovery plans for listed species (Carroll et al. 2006). Therefore, proactive adoption of this strategy might preclude the need to list additional native trout as threatened or endangered species.

Saving native trout diversity requires the conservation of genetically pure populations, protection and restoration of life history diversity, and protection of populations across the historic range. Protecting larger, stronghold populations is important because such populations will have a better chance of surviving future disturbances, including those associated with climate change. Redundancy among these elements is important given that many populations are small, occupy reduced habitat in fragmented stream systems, and therefore are vulnerable to extirpation. Table 1 provides a more complete description of the application of the 3-R strategy to native trout.

The long-term persistence of populations is likely to require maintenance of larger population sizes and habitat patches than currently exist for many populations. Hilderbrand and Kershner (2000) and Rieman et al. (2007) have provided quantifiable criteria for measuring the likelihood of long-term persistence of trout populations. At a minimum, an effective population size of 500 interbreeding adults is necessary, which equates to a census population size of approximately 2,500 individuals greater than or equal to 75 mm total length (Hilderbrand and Kershner 2000). Many existing populations of native

Table 1. Applying the 3-R strategy to develop goals, objectives, and indicators of success in the conservation of native trout.

Management Goal	Objectives	Indicators of Success
Representation	1. Conservation of genetic diversity 2. Protection and restoration of life history diversity 3. Protection of geographic (ecological) diversity	1. Presence of genetically pure populations 2. Presence of all life histories that were present historically 3. Presence of peripheral populations
Resilience	1. Protect/restore strongholds 2. Protect/restore metapopulations	1. Occupied stream habitat exceeds 27.8 km and habitat patch size exceeds 10,000 ha 2. Occupied stream habitat supports migratory life history and exceeds 50 km and habitat patch size exceeds 50,000 ha
Redundancy	1. Protect multiple populations within each sub-basin	1a. 5 persistent populations within each sub-basin, or 1b. 1 or more larger strongholds within each sub-basin, or 1c. 1 metapopulation within each larger basin

western trout, especially those located in the periphery of their ranges, fail to meet this threshold (Haak et al. 2010a).

Analytical Approach for Case Studies

To describe and map the existing conservation portfolio, we use population data from the Yellowstone cutthroat trout (May et al. 2007) and Rio Grande cutthroat trout (Alves et al. 2007) range-wide status assessments. Supplemental analyses of population persistence and the distribution of remaining peripheral populations further inform our assessment (Haak et al. 2010a). In addition to restoring and protecting representative populations, we quantify the need for resilience and redundancy by following the recommendation of Rieman et al. (2007), who called for a minimum of five persistent populations within each sub-basin (4th code hydrologic unit), or one or more strongholds, or one metapopulation if sufficient habitat is available. A stronghold population is defined as one occupying at least 27.8 km of interconnected stream habitat and a habitat patch size of 10,000 ha, while a metapopulation occupies at least 50 km of interconnected stream habitat and a habitat patch size of 50,000 ha and supports a migratory life history (Dunham and Rieman 1999; Hilderbrand and Kershner 2000).

Following are two case studies that apply the 3-R strategy to Yellowstone cutthroat trout and Rio Grande cutthroat trout. Although our analyses use the most recent range-wide population and habitat data available, local stream conditions may exist that are not reflected in our assessment. Therefore, it is important that the results of our analyses are considered in conjunction with more detailed knowledge of current, local conditions to devise the most appropriate strategy.

Yellowstone Cutthroat Trout Case Study

Conservation Status

In the most recent status review, May et al. (2007) identified 383 conservation populations of Yellowstone cutthroat trout occupying 11,600 km of stream habitat and 66,600 ha of lake habitat. This encompasses 54% of historically occupied sub-watersheds. The majority of habitat lost has been in the southwest and east portions of the range as well as the lower elevations whereas populations in the central core, located primarily on protected federal lands, have remained relatively intact. However, these populations are increasingly at risk from the more ubiquitous threats of nonnative species and climate change.

Nonnative salmonids threaten the persistence of Yellowstone cutthroat trout primarily through hybridization and predation. Predation is particularly problematic in Yellowstone Lake, where an illegal introduction of lake trout *Salvelinus namaycush* has led to a decline in the abundance of the native cutthroat trout in the lake and spawning tributaries (Gresswell et al. 1994). May et al. (2007) estimated that about 28% of the historic range currently supports genetically unaltered populations with the greatest declines occurring in the lower Snake River drainage. An increased management emphasis on protecting genetic integrity by isolating populations above barriers has been at the expense of migratory life histories, the cornerstone of evolution and adaptation within the subspecies.

Portfolio Status and Recommendations

The results of our 3-R strategy show a conservation portfolio for Yellowstone cutthroat trout that, at a range-wide scale, supports all the desired conservation elements: 45% of the occupied stream habitat supports genetically pure cutthroat, 54% supports a migratory life history, and 5% contributes to the subspecies' geographic diversity. However, when these same results are analyzed at the population level, the portfolio appears less secure as the inherent vulnerability of many of these small, isolated populations becomes evident. Reduced resilience and loss of genetic integrity may exacerbate the effects of a rapidly changing environment. Table 2 provides a population level summary of these factors.

Table 2 highlights the loss of resilience that has occurred across much of Yellowstone cutthroat trout's historic range, particularly downstream of the central core in the headwaters of the Yellowstone and Snake river basins. Nineteen of the 28 migratory populations are classified as resilient. The remaining populations with migratory life histories are associated with small lakes and tributary habitats that support resident and adfluvial life histories. The loss of resilience is particularly pronounced in the populations representing genetic and geographic diversity in which just over 10% are considered resilient.

Given the limited size of many of these populations, it is not surprising that our assessment of redundancy at the sub-basin scale found that only 22 of the 37 sub-basins (60%) within the historic range meet our portfolio objectives. The majority of these are concentrated around the core populations at the center of the range while the sub-basins around the periphery contain few strongholds and existing populations have limited extents. One notable exception is the 153-km peripheral metapopulation in Goose Creek at the extreme southwest extent of the range. At the basin-scale, our goal of restoring one large interconnected metapopulation within each major river basin has been met in all of the basins with the exception of the Tongue River basin, where it is unlikely that a metapopulation could ever be restored to this river basin.

An additional threat to the long-term persistence of Yellowstone cutthroat trout is loss of genetic integrity through hybridization with nonnative trout. As Table 2 indicates, slightly less than one-half (28 out of 57) of the migratory populations are genetically pure, but they occupy just 30% (1,866 km out of 6,270 km) of the stream habitat occupied by migratory populations and only seven populations are resilient. Peripheral populations important to geographic diversity have maintained a higher degree of genetic integrity (84% are genetically pure), but they also lack resilience with only one stronghold and no metapopulations.

Although the range-wide portfolio for this subspecies contains a diversity of important elements, our analysis suggests that further work is needed to secure these elements. Increasing reliance on small, isolated populations increases the vulnerability of the entire portfolio. Larger populations that are genetically pure and support a migratory life history are critical to the success of the portfolio and should be given a higher priority. Removal of nonnative trout that threaten resilient and migratory populations should also be a priority so that these larger populations become part of the core. The smaller genetically pure populations that exist at lower elevations and around the margins of the range, are essential to maintaining the genetic and geographic diversity of the portfolio and should be improved where possible and protected. In these sub-basins, it is important to extend and reconnect the smaller populations when possible and otherwise establish multiple populations in different watersheds within a sub-basin to guard against loss from some form of disturbance.

Rio Grande Cutthroat Trout Case Study

Conservation Status

Rio Grande cutthroat trout have declined across their range and now occupy slightly more than 10% of their historic stream habitat in Colorado and New Mexico. They have been eliminated from the Texas

Table 2. Population summary of 3-R strategy for stream-dwelling Yellowstone cutthroat trout. Populations supporting more than one conservation element (e.g. genetically pure and migratory life history) are counted in each column so that totals for representation elements may exceed range-wide population counts reported in the first column. Genetically pure populations are analyzed in more detail to quantify the overlap between genetic integrity and the other two elements of representation.

			Representation (All Populations n=306)			Genetically Pure (Populations n=228)	
		Range-wide	Genetically Pure	Migratory Life History	Geog. Diversity	Migratory Life History	Geog. Diversity
	Total Pops.	306	228	57	38	28	32
Resiliency	Stronghold	42	19	6	3	0	1
	Metapops.	19	7	19	1	7	0
Redundancy	Persistent	109	60	35	6	11	5
	Total Km	11,600	5,328	6,270	668	1,866	286

portions of their range. Much of the decline can be attributed to habitat degradation from livestock grazing, timber harvest, and water diversions as well as competition and genetic introgression from introduced rainbow trout and nonnative cutthroat trout. In an effort to protect populations from nonnative species, many populations have been isolated above in-stream barriers. Alves et al. (2007) did not identify any metapopulations or even strongly networked populations in the 2006 status assessment.

Recognizing the need to establish an interconnected stronghold population of Rio Grande cutthroat trout, the Rio Costilla watershed restoration project was undertaken as a collaborative effort among federal, state, and private interests. It is an ambitious 10-year effort to establish a genetically pure metapopulation of Rio Grande cutthroat trout in 240 km of interconnected streams and 25 lakes within this northern New Mexico watershed. Once established, the metapopulation will support the full range of life history types of the native cutthroat trout as well as other native fishes.

Portfolio Status and Recommendations

The existing management portfolio for Rio Grande cutthroat trout is at higher risk and supports less diversity than does the portfolio for Yellowstone cutthroat trout. From a range-wide perspective, genetic integrity of Rio Grande cutthroat trout is the strongest of the portfolio elements with genetically pure populations occurring in more than 75% of the occupied habitat, whereas life history diversity is meager. Peripheral populations representing geographic diversity occupy just 6% of the occupied habitat compared to 10% historically. All historically disjunct populations are extinct and just over 15% of continuous peripheral populations remain (Haak et al. 2010a).

Table 3 summarizes the results of the 3-R strategy at the population level and highlights the vulnerability of remaining populations. With no metapopulations and only one stronghold population, the subspecies has little resilience to environmental change. None of the four migratory populations include a fluvial life history form; instead, these populations migrate between tributary streams and small lakes. Given the lack of resilience and resultant risk of population extinctions due to environmental disturbances, redundancy is particularly important to the long-term persistence of Rio Grande cutthroat trout. However, only two of the 19 historically occupied sub-basins in New Mexico and Colorado meet the goal of five persistent populations. In five sub-basins, all populations are extinct and four do not contain any persistent populations. Our goal of one large metapopulation in each major river basin has also not been met, but the Rio Costilla restoration project will satisfy this objective for the lower Rio Grande.

Reducing the risk and increasing the diversity for the Rio Grande cutthroat trout portfolio requires the expansion of some existing populations to improve redundancy of genetically pure populations. This is particularly important for the few remaining peripheral populations to preserve geographic diversity. Restoring resilience and a fluvial life history by reconnecting isolated populations will require a long-term commitment and the need to address the threat to genetic integrity of nonnative trout. Opportunities exist in the Pecos River basin and Rio Grande headwaters to reestablish a metapopulation in each basin and restore some resilience to the

Table 3. Population summary of 3-R strategy for stream-dwelling Rio Grande cutthroat trout. Populations supporting more than one conservation element (e.g. genetically pure and migratory life history) are counted in each column that applies so totals for representation elements may exceed range-wide population counts reported in the first column. Genetically pure populations are analyzed in more detail to quantify the overlap between genetic integrity and the other two elements of representation.

			Representation (All Populations n=121)			Genetically Pure (Populations n=89)	
		Range-wide	Genetically Pure	Migratory Life History	Geog. Diversity	Migratory Life History	Geog. Diversity
	Total Pops.	121	89	4	8	4	4
Resiliency	Stronghold	1	1	0	0	0	0
	Metapop.	0	0	0	0	0	0
Redundancy	Persistent	26	20	2	2	2	0
	Total Km	1,124	872	64	68	64	22

portfolio. However, the opportunities in the Upper Canadian River basin are more limited; hence, restoration actions will need to focus on redundancy and ensuring that multiple examples of these populations exist throughout the basin to guard against basin-wide extirpation.

ACKNOWLEDGEMENTS

Discussions with Jeff Kershner (USGS), Bob Gresswell (USGS), Helen Neville (TU), Dan Dauwalter (TU), and Kurt Fesenmyer (TU) assisted our analysis and manuscript preparation. This research was supported by U.S. Geological Survey's Global Change Research Program and TU's Coldwater Conservation Fund.

REFERENCES

Alves, J. E. et al. 2007. Range-wide status of Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*), 2007. Rio Grande Cutthroat Trout Conservation Team, Colorado Division of Wildlife, Denver.

Brown, D. K., A. A. Echelle, D. L. Propst, J. E. Brooks, and W. L. Fisher. 2001. Catastrophic wildfire and number of populations as factors influencing risk of extinction for Gila trout (*Oncorhynchus gilae*). Western North American Naturalist 61(2):139-148.

Carroll, C., M. K. Phillips, C. A Lopez-Gonzalez, and N. H. Schumaker. 2006. Defining recovery goals and strategies for endangered species: the wolf as a case study. BioScience 56(1):25-37.

Clark, G. M. 2010. Changes in patterns of streamflow from unregulated watersheds in Idaho, western Wyoming, and northern Nevada. Journal of the American Water Resources Association 46:486-497.

Colyer, W. T., J. L. Kershner, and R. H. Hilderbrand. 2005. Movements of fluvial Bonneville cutthroat trout in the Thomas Fork of the Bear River, Idaho-Wyoming. North American Journal of Fisheries Management 25:954-963.

Fausch, K. D., B. E. Rieman, M. K. Young, and J. B. Dunham. 2006. Strategies for conserving native salmonid populations at risk from nonnative fish invasions: tradeoffs in using barriers to upstream movement. U. S. Forest Service General Technical Report RMRS-GTR-174.

Figge, F. 2004. Bio-folio: applying portfolio theory to biodiversity. Biological Conservation 13:827-849.

Gresswell, R. E., W. J. Liss, and G. L. Larson. 1994. Life-history organization of Yellowstone cutthroat trout (*Oncorhynchus clarkii bouvieri*) in Yellowstone Lake. Canadian Journal of Fisheries and Aquatic Sciences 51(Supplement 1):298-309.

Haak, A. L., J. E. Williams, H. M. Neville, D. C. Dauwalter, and W. T. Colyer. 2010a. Conserving peripheral trout populations: the values and risks of life on the edge. Fisheries 35:in press.

Haak, A. L., J. E. Williams, D. Isaak, A. Todd. C. Muhlfeld, J. L. Kershner, R. Gresswell, S. Hostetler, and H. M. Neville. 2010b. The potential influence of changing climate on persistence of native western salmonids. U.S. Geological Survey Open-File Report: in press.

Hilderbrand, R. H. and J. L. Kershner. 2000. Conserving inland cutthroat trout in small streams: how much habitat is enough? North American Journal of Fisheries Management 20:513-520.

Luce, C. H., and Z. A. Holden. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948-2006. Geophysical Research Letters. Vol. 36, L16401, doi:10.1029/2009GL039407.

May, B. E., S. E. Albeke, and T. Horton. 2007. Range-wide status assessment for Yellowstone cutthroat trout (*Oncorhynchus clarkia bouvieri*): 2006. Yellowstone Cutthroat Trout Interagency Coordination Group, Helena, Montana.

Neville, H., J. Dunham, A. Rosenberger, J. Umek, and B. Nelson. 2009. Influences of wildfire, habitat size, and connectivity on trout in headwater streams revealed by patterns of genetic diversity. Transactions of the American Fisheries Society 138:1314-1327.

Poff, N. L. 2002. Ecological response to and management of increased flooding caused by climate change. Philosophical Transactions Royal Society of London 360:1497-1510.

Primack, R. B. 2002. Essentials of conservation biology, 3rd edition. Sinauer Associates Publishers, Sunderland, Massachusetts.

Rahel, F. J., and J. D. Olden. 2008. Assessing the effects of climate change on aquatic invasive species. Conservation Biology 22:521-533.

Rieman, B. E., and J. B. Dunham. 2000. Metapopulations and salmonids: a synthesis of life history patterns and empirical observations. Ecology of Freshwater Fish 9:51-64.

Rieman, B. E., D. Isaak, S. Adams, D. Horan, D. Nagel, C. Luce, and D. Myers. 2007. Anticipated climate warming effects on bull trout habitats and populations across the Interior Columbia River Basin. Transactions of the American Fisheries Society 136:1552-1565.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465: 609-612.

Shaffer, M. L., and B. A. Stein. 2000. Safeguarding our precious heritage. Pages 301-321 in, B. A. Stein, L. S. Kutner, and J. S. Adams, eds. Precious heritage: the status of biodiversity in the United States. Oxford University Press, New York.

Williams, J. E., A. L. Haak, H. M. Neville, and W. T. Colyer. 2009. Potential consequences of climate change to persistence of cutthroat trout populations. North American Journal of Fisheries Management 29:533-548.

LIFE HISTORY DIVERSITY AND ENVIRONMENTAL DISTURBANCE IN NATIVE TROUT POPULATIONS: A HEURISTIC MODEL OF STRUCTURE AND SYNCHRONIZATION OF DISTURBANCE REGIMES

Mike Ebinger [1], Warren Colyer [2*], and Kirk Dahle [3]
[1] Big Sky Institute, Montana State University, Bozeman, Montana
[2] Trout Unlimited, Director - Watershed Programs, Missoula, Montana
[3] Trout Unlimited, Coordinator - Bear River Native Trout Program, Logan, Utah
[*]Corresponding Author

ABSTRACT — Life history diversity historically buffered cutthroat trout *Oncorhynchus clarkii* populations against environmental extremes. However, anthropogenic activities during the last century have imposed selective pressures against migratory life histories, and many cutthroat trout populations now are isolated in headwater systems and express only resident life history strategies. This reduction in life history diversity, coupled with projections for an increasingly variable climate, may compromise the future viability of these populations. In this paper we explored the structure of the environmental variation associated with climate change, including how it might impact cutthroat trout populations with and without migratory life history components. We adapted a population model developed by Van Kirk and Hill (2007) to incorporate varying levels of autocorrelation within three types of disturbance that are predicted to increase in frequency and magnitude as a result of climate change—drought, flood, and fire. We found that disturbance regimes that included autocorrelation reduced hypothetical trout populations much more than those that did not. However, across all disturbance scenarios migratory fish often were able to sustain their life history when the resident portion of the population went extinct. These results suggest that conservation strategies designed to restore migratory life histories can build resiliency in cutthroat trout populations.

INTRODUCTION

North American salmonid species have faced formidable environmental changes at multiple spatial and temporal scales during their long evolutionary history (Waples et al. 2008). Repeated glacial advances and retreats, unpredictable and landscape-altering floods, massive fires, and prolonged droughts are among a few of the factors that impacted aquatic habitats and favored the development of alternative life history strategies (Watry and Scarnecchia 2008). Resulting life history diversity within populations buffered them against these types of environmental extremes by spreading risk across habitats at greater spatial and temporal scales (Northcote 1997; Hilborn et al. 2003; Koski 2009). Today, where life history diversity remains intact, we see the benefit of these adaptations as migratory fish continue to buffer native trout populations against year-class failures following environmental disturbances (Rieman and Clayton 1997). In the case of catastrophic events, low rates of migratory straying provide a source of colonists that can promote recovery after local population declines or extinctions (Leider 1989; Northcote 1997; Dunham et al. 1997; Waples et al. 2009).

Our understanding of the adaptive advantages of salmonid life history diversity and their importance to individual- and meta-populations has been dominated by the study of anadromous species. This picture now may be changing, as potamodromy—migration solely within fresh water—is increasingly recognized as a critical development in the adaptation of salmonids to a diverse suite of aquatic environments (Northcote 1997). The colonization of such diverse habitats probably required a fluvial life history, and we can infer that historical cutthroat trout *Oncorhynchus clarkii* populations contained both resident and migratory individuals. In fact, contemporary cutthroat populations with access to large main stem systems often exhibit both resident and fluvial components (Schmetterling 2001, 2003; Colyer et al. 2005). Unfortunately, anthropogenic activities during the last century have imposed selective pressures against migratory life histories by fragmenting habitats and blocking migration corridors (Thurow et al. 1988; Rieman and McIntyre 1993). As a result, many western native trout populations now comprise only isolated, resident populations in headwater systems (Young 1995). This widespread reduction in life history diversity, coupled with projections for an increasingly variable and unpredictable climate, are likely to compromise the future viability of many cutthroat trout populations.

Ecologists recognize that environmental variation can significantly impact individual reproductive success and mortality, and ultimately population dynamics (Andrewartha and Birch 1954; Laakso et al. 2003). To date, most theoretical work on environmental variability (i.e. noise) has assumed an absence of autocorrelation, partly for want of a more general model of noise, and partly for want of guidance from data (Vasseur & Yodiz 2004). More recently, however, ecology has seen a growing interest in the temporal structure of environmental noise—specifically, in its degree of autocorrelation—as scientists are beginning to recognize that many environmental fluctuations (i.e. temperature and rainfall) often are positively auto-correlated (Mandelbrot and Wallis 1969; Steele 1985; Gonzalez and Holt 2002). It has been posited that climate change could increase this autocorrelation in environmental variables (Wigley et al. 1998), and positively auto-correlated environments could result in longer sequences of favorable or unfavorable conditions (Wilmers et al. 2007).

Autocorrelation structure in noise often is represented by a signal's spectral density, with different types of noise named after different colors. As such, the absence of autocorrelation is portrayed as "white noise," which assumes that the values of a random signal at two instants in time are completely independent of each other (Halley 1996). Alternatively, "pink" and "red" noise represent increasing levels of temporal autocorrelation within environmental noise, and can be created by "coloring" or "dyeing" a white noise signal. From an ecological perspective this coloring of the noise signal allows for the storage or buffering of chance events over time by the physical environment or biological processes (Sabo and Post 2008). Correlations among environmental events allow perturbations to have longer-lasting effects and arguably provide "a better null model for environmental variability" (Halley 1996).

Williams et al. (2009) indentified three specific types of environmental perturbations that are predicted to increase as a result of global climate change and likely will impact cutthroat trout populations: warmer summer temperatures, winter flooding, and wildfire. Specifically, in the Intermountain West, we expect more instances of prolonged drought (IPCC 2007), longer fire seasons with larger, more intense wildfires (Westerling et al. 2006), and uncharacteristic winter floods following more frequent rain on snow events (Poff 2002). Given these environmental risk factors, we were interested in how their temporal structures interact with and influence population dynamics for both resident and migratory life history components within cutthroat trout populations. Our objective was to explore these three primary environmental threats (fire, drought, and winter floods) by explicitly incorporating different temporal structures into appropriate vital rates for a simulated population of cutthroat trout. We investigate three specific questions: (1) how the autocorrelation structure (i.e. color of noise) of different disturbance scenarios influences the frequency, strength, and duration of population trends, (2) how synchronized disturbances and catastrophic events act synergistically to alter population trajectories, and (3) to what degree migratory life history strategies buffer resident populations against local extinctions.

METHODS

Population Model

We developed a stochastic (environmental) population model following the basic structure and parameterization of Van Kirk and Hill (2007), which was based on cutthroat trout in the upper Snake River basin. Due to space limitations here, we refer readers to Van Kirk and Hill (2007) for model parameters, model flowcharts, model scaling, and density dependence specifications. The model included both life-stage and age-class mechanics, and only the female segment of the population. We simulated both migratory and resident life histories, using a very simplified structure where we controlled the proportion of migratory and resident juvenile carrying capacity within the system assuming a 1:1 ratio. We calculated abundances for each stage (eggs, emergent fry, pre-winter juveniles, age-1 subadults) and adult age-class (see Figure 1 in Van Kirk and Hill 2007). We calculated transitions between life stages by multiplying life-stage and age-class abundance by the appropriate survival rate (see Table 1 in Van Kirk and Hill 2007). Because we assumed that most juvenile trout mortality occurs as fish compete for limited habitat during the winter (Mitro and Zale 2002; Mitro et al. 2003; Van Kirk and Hill 2007), we modeled juvenile winter survival as a density-dependent process whereby the number of surviving individuals approaches carrying capacity asymptotically as pre-winter juvenile abundance increases (Van Kirk and Hill 2007). We assumed that after their first winter fish survive at annual rates that are independent of density. All other survival rates were constant with respect to abundance. Though built entirely upon Van Kirk and Hill (2007), our model is fundamentally different in the way we incorporate environmental stochasticity. Van Kirk and Hill (2007) used a white-noise structure to simulate environmental stochasticity, while we use a suite of "colored noises" that vary from white to pink to red (i.e. no autocorrelation to some to strong autocorrelation). Annual vital rates (typically survival) were adjusted by multiplying the vital rate by the appropriate standardized (mean=0, variance=1) colored noise to model the impact of different disturbance regimes.

Disturbance Regimes and Environmental Stochasticity

We considered each of our three environmental disturbances independently, modeling winter flood and drought as continuous annual adjustments to vital rates as described above. We modeled fire as a onetime catastrophic event due to the limited spatial (one population) and temporal (100 years) scales of our model relative to fire return intervals in the western United States. For winter flood, drought, and main stem conditions (i.e. environmental stochasticity that migratory adults are subjected to) we initially created unique time-series of white noise to serve as stochastic variation for each risk. This initial white-noise signal was created by generating 100 random numbers (one for each year) from a uniform distribution [0 1] with a mean of 0 and a standard deviation of 1. We repeated this process 100 times to produce a 100 x 100 (years x replications) array of white-noise structure for all three event types (e.g. flood, drought, and main stem conditions). We then "dyed" (i.e. filtered) this white noise, using a scale-invariant power spectrum to produce different types of $1/f^{\beta}$ noise (Halley 1996). The spectral density of the data is proportional to f^{β}, where f is the frequency and β is the spectral exponent (β =0 is white noise, β =1 is pink noise, and β =2 is red noise; Figure 1). We varied β in increments of 0.25 from white noise to red noise, creating nine unique categories of autocorrelation built from the same underlying uniform distribution. We standardized each filtered time series to have a mean of 0 and unit variance (Morales 1999). This standardization ensured that the means and variances of the different colors of noise were equal for the 100-year simulation period and the only difference was the temporal autocorrelation structure. To convert these noise signatures to population vital rates, we multiplied the colored time series by the standard deviation of the appropriate fixed vital rates, and added that value to the mean of the vital rate. We outline the individual disturbances below and our assumptions about their impacts on cutthroat trout vital rates.

Drought

Prolonged and intensified drought conditions are expected across much of the western half of North America and likely will impact native trout populations. In arid western regions, snowpack is projected to decrease as more winter precipitation falls as rain, and, as a result, many streams will experience prolonged periods of high temperatures and low flows during summer months (IPCC 2007). Our population model assumed that adult migratory fish were able to circumvent the high temperatures and low flows during summer months by using main stem river migration corridors to seek out thermal refugia. Accordingly, we considered the effects of drought to primarily impact resident fish and the subset of juvenile fluvial fish that had not yet migrated out of smaller natal tributaries (i.e. age < 3). We assumed that the primary impact was on survival and multiplied age-specific survival rates by the appropriate environmental noise series to simulate the effects of drought over the 100-year simulation period. We also ran a subset of models with the overwinter juvenile carrying capacity (K) varied according to the same autocorrelation structure as drought (Table 1 – Scenarios A,C,&E).

Winter Flood

In addition to lower summer flows associated with more frequent and more severe drought conditions, winter hydrology also is projected to change in the Intermountain West. Warmer winter air temperatures will lead to more frequent rain-on-snow events, which cause uncharacteristic winter floods (Hamlet and Lettenmaier 2007; Williams et al. 2009). Winter floods have been historically uncommon in the Intermountain West, but Erman et al. (1998) argue that in California streams midwinter floods cause more intense bed scour, because the accumulated snowpack prevents streams from overtopping and gaining the floodplain as they are able to do in the spring. Flood events like these can impact adult trout but are most detrimental for juvenile trout, which can be displaced and even killed by flood flows (Seegrist and Gard 1972). We assumed these impacts from winter floods would primarily influence juvenile survival for both migratory and resident life histories by creating a mismatch with the hydrological conditions to which they are adapted (Fausch et al. 2001; Poff 2002). We also ran a subset of models where the juvenile winter capacity of the environment was altered according to the winter flood noise time series (Table 1 – Scenarios B,D,& F).

Fire

Fires cause both direct mortality of trout (probably due to temperature spikes) and indirect, long-term negative impacts by reducing overhead cover, elevating water temperatures, and increasing sediment inputs into streams (Gresswell 1999; Dunham et al. 2003 and references therein). However, because we are modeling a single population, the typical length of fire return intervals did not allow for manipulation of vital rates with the colored environmental noise approach used for drought and winter floods. Instead we simulated a single fire event and varied its intensity (relatively low impact to near devastation). To investigate the impact of synchronized disturbances, we varied the timing of a fire event starting in year 25 and ending in year 50. This generated 25 different simulation outputs for a set of fire, flood, and drought conditions. Similar to drought, we considered fire to have impacts on all non-migrating fish, including immature fluvial life histories. We modeled the impacts of fire as percent reductions in survival rates for the year of the fire and varied this from 25% (low intensity fire) to 99% (high intensity fire).

Main Stem Conditions

One of the assumptions of our conceptual framework is that adult migratory fish are subjected to different environmental conditions when they are in large main stem rivers compared to resident fish and immature migratory fish in tributary habitats. Accordingly, we modeled the environmental stochasticity driving the survival of adults in the main stem river separately from the environmental stochasticity of non-main stem individuals. We first allowed main stem conditions to vary according to β (Table 1 – Scenarios A and B). We then fixed main stem conditions as random white noise to capture the notion that larger habitats and migration corridors can act as, or allow access to, refugia from drought, flood, and fire events that may have greater impact in lower order tributaries (β=0; Table 1 – Scenarios C-F).

Model Metrics

To evaluate impacts to the population we summarize the proportion of simulations where the population decreased to below a threshold value φ. For this paper we present the results when φ equals 20% of the deterministic equilibrium discussed above (i.e. an 80% reduction in population size). We considered this to be a major reduction in fish numbers that would warrant conservation and managerial caution, though we do concede that this level is arbitrary. While our model does not allow individuals to switch from resident to migratory life history or vice-versa, we use the persistence of the migratory life history when the resident population goes extinct as a proxy for the migratory life history “buffering” the system from perturbations. To ensure that we quantified actual persistence in the migratory life history, we restricted our analysis to resident extinctions occurring before year 80, therefore requiring 20 years of persistence in the migratory life history.

RESULTS

The degree of autocorrelation (i.e. color of noise) had a strong effect on the proportion of simulations falling below the threshold level (φ), which represented an 80% reduction from the deterministic carrying capacity. Regardless of the disturbance scenario modeled, "whiter" noises (i.e. β=0,0.25) showed no drops below φ for any simulations (Figure 2). In general, the proportion of simulations falling below φ during the 100-year simulation increased with increasing autocorrelation (i.e. 'redness') following a sigmoidal pattern (Figure 2). The proportion of simulations falling below φ also was sensitive to the way juvenile carrying capacity (K) was modeled. Allowing K to vary according to the same noise structure as drought (e.g. models a, c, & e) increased effect sizes slightly compared to the same disturbance type when modeling K as a function of the noise structure used for winter flooding (models b, d, & e).

The proportion of life-history extinction events (i.e. either residents or migratory component of the population goes extinct) was strongly influenced by autocorrelation structure of environmental stochasticity and disturbance scenario (Figure 3). Neither life history experienced extinction events during any simulations under white noise. Instead, extinctions did not appear until reaching pink levels above (i.e. β ≥ 1), with the maximum proportion (0.22) occurring under red noise. The disturbance scenario also affected the extinction rate, with the most pronounced effects in the migratory life history. When the main stem conditions were considered to always vary under a white noise structure (β=0; Scenarios C,D,E,& F) there were significant declines compared to conditions when main stem noise fluctuated according to β (scenarios A & B). Further, when winter flood stochasticity was fixed as white noise (scenarios E & F) there were very few migratory extinction events (maximum of 7% under scenario F at β = 2.0). Resident life-history extinction rates were identical for scenarios (A & C) and (B & D) where the only differences were how environmental stochasticity was modeled for migratory fish, which were assumed to be in the main stem river.

Across all noise structures (β=0 to β=2) migratory fish often were able to sustain their life history when the resident portion of the population went extinct (Figure 4). Whereas we did not model it explicitly, we assumed that under this scenario migratory individuals would refound, or "save," the resident population. The proportion of resident extinctions "saved" was relatively even under disturbance scenarios A and B, where the main stem fluctuated according to β, and over 40% of the resident extinctions before year 80 were associated with persistent migratory components. The persistent modeling of main stem environmental stochasticity as white noise (C – F) increased this proportion. The greatest impact of migratory life history relative to resident extinctions occurred with main stem and winter flood stochasticity modeled as persistent white noise (E & F), where more than 85% of the resident extinctions (on average) were associated with the persistence of the migratory component.

Fire intensities with vital rate reductions less than 75% had negligible impact on the proportion of simulations falling below φ. However, at fire intensities of 75% and above we were able to detect differences in the proportion of simulations falling below φ and the frequency of extinction events (for resident life-histories). For all levels of fire intensity the proportion of simulations falling below φ increased with redness of noise (β). The addition of high intensity catastrophic fire increased the frequency of extinction events for resident fish, and, at the highest levels (β = 2 and fire intensity = 95% reduction of resident survival) there was a nearly 3 fold increase in resident extinction rates. Adult migratory fish (age > 2) were assumed to be out of the system during these fire events and therefore fire had marginal to no impact on migratory extinction rates, as it only impacted premature migratory individuals.

DISCUSSION

Our results suggest that both the structure of environmental variation and the extent of life history diversity within cutthroat trout populations are likely to affect how they respond to climate change. To date the status quo for incorporating environmental stochasticity into population models has been to assume random variation (i.e. 'white noise'; Lawton 1997). However, it seems increasingly likely that real environmental variation—especially that which is projected by many climate change models—is much different from white noise (Halley 1996). Instead, the potential for climate change to manifest itself in longer-lasting sequences of favorable or unfavorable conditions (Houghton et al. 2001; Wilmers et al. 2007) means environmental perturbations will be more likely to occur simultaneously, resulting in increasingly violent population fluctuations. In our model these synchronized disturbance scenarios, or "perfect storms," appeared more often with increasing autocorrelation and pushed both resident and migratory life-histories toward extinction with increasing frequency as they increased in strength and/or duration. However, the differential environmental pressure on migratory and resident life histories provided a mechanism to buffer these events, and the migratory life history was able to persist during 41% to 92% of the resident extinction events, depending on the modeling scenario (Figure 4).

This resiliency of migratory life histories in the face of environmental disturbance depended on the assumption that main stem habitats were less prone than tributary habitats to experience synchronized disturbances. This assumption was based on the idea that larger main stem habitats were more likely to provide refugia from warm water temperatures (e.g. deep pools, ground water return), floods, and fires than lower order tributary habitats, and that main stem inhabitants were able to move along those corridors to access habitats that were not experiencing synchronized disturbances (e.g. other tributaries).

The introduction of fire as a onetime catastrophic perturbation increased the frequency of population reductions (and fluctuations). The fact that our trout populations were robust to fire intensities below 75% reductions in survival suggests that only truly catastrophic fires (i.e. those that reduced population by >75%) elicited a population level response. The fact that populations in our model were able to rebound from a 95% reduction in resident survival is a testament to the resiliency of salmonids in general. These results are consistent with empirical data suggesting that over the short term trout have the ability to adapt to and recover from changes in their environment associated with intense wildfire (Rieman et al. 1997). However, when we modeled fire intensity as reducing our populations by 75% or more, the impacts to the resident populations were devastating. Those populations experienced nearly a threefold increase in extinctions given high intensity fire and highly auto-correlated flood and drought regimes ($\beta = 1$; i.e. red noise). Thus, our conclusions about resiliency, as well as those of Rieman et al. (1997), seem to require some level of life history diversity within the system.

We believe that our conceptual approach offers a compelling framework for examining both the structure of environmental disturbance regimes that western native trout are likely to face in the coming decades and the impacts that these perturbations are likely to have on populations with and without migratory life histories. Our results suggest that conservation strategies designed to restore and protect life history diversity will be among the most effective strategies to buffer cutthroat trout populations against climate change.

LITERATURE CITED

Andrewartha, H. G., and L. C. Birch. 1954. The distribution and abundance of animals. The University of Chicago Press, Chicago, Illinois.

Colyer, W. T., J. L. Kershner, and R. H. Hilderbrand. 2005. Movements of fluvial Bonneville cutthroat trout in the Thomas Fork of the Bear River, Idaho–Wyoming .North American Journal of Fisheries Management 25:954-963.

Dunham, J. B., G. L. Vinyard, and B. E. Rieman. 1997. Habitat fragmentation and extinction risk of Lahontan cutthroat trout. North American Journal of Fisheries Management 17:910–917.

Dunham, J. B., M. K. Young, R. E. Gresswell, and B. E. Rieman. 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. Forest Ecology and Management 178:183-196.

Erman, D. C., E. D. Andrews, and M. Yoder-Williams. 1988. Effects of winter floods on fisher in the Sierra Nevada. Canadian Journal of Fisheries and Aquatic Sciences 45:2195-2200.

Fausch, K. D., Y. Taniguchi, S. Nakano, G. D. Grossman, and C. R. Townsend. 2001. Flood disturbance regimes influence rainbow trout invasion success among five Holarctic regions. Ecological Applications 11:1438-1455.

Gonzalez, A., and R. D. Holt. 2002. The inflationary effects of environmental fluctuations in source-sink systems. Proceedings of the National Academy of Sciences, USA, 99:14872-14877.

Gresswell, R. E. 1999. Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society 128:193-221.

Halley, J. M. 1996. Ecology, evolution and 1/f noise. Trends in Ecology and Evolution 11:33-37.

Hamlet, A. F., and D. P. Lettenmaier. 2007. Effects of 20th century warming and climate variability on flood risk in the western U.S. Water Resources Research 43:W06427.

Hilborn, R., T. P. Quinn, D. E. Schneider, and D. E. Rogers. 2003. Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences, USA, 100:6564-6568.

Houghton, J. T., Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, and D. Xiaosu, Eds. 2001. Climate Change 2001: The Scientific Basis: Contributions of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

IPCC (Intergovernmental Panel on Climate Change). 2007. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, Pachauri, R.K and Reisinger, A. (eds.)]. IPCC, Geneva, Switzerland.

Koski, K. 2009. The fate of Coho salmon nomads: the story of an estuarine-rearing strategy promoting resilience. Ecology and Society 14(1):4.

Laakso, J., V. Kaitala, and E. Ranta. 2003. Non-linear biological responses to disturbance: consequences on population dynamics. Ecological Modeling 162:247-258.

Lawton, J. H. 1997. The science and non-science of conservation biology. Oikos 79:3-5.

Leider, S. A. 1989. Increased straying by adult steelhead trout, *Salmo gairdneri*, following the 1980 eruption of Mt. St. Helens. Environmental Biology of Fishes 24:219-229.

Mandelbrot, B. B., and J. R. Wallis. 1969. Some long-run properties of geophysical records. Water Resources Research 5:321-340.

Mitro, M. G., and A. V. Zale. 2002. Seasonal survival, movement, and habitat use of age-0 rainbow trout in the Henry's Fork of the Snake River, Idaho. Transactions of the American Fisheries Society 131:271-286.

Mitro, M. G., A. V. Zale and B. A. Rich. 2003. The relation between age-0 rainbow trout (*Oncorhynchus mykiss*) abundance and winter discharge in a regulated river. Canadian Journal of Fisheries and Aquatic Sciences 60:135–139.

Morales, J. M. 1999. Viability in a pink environment: why "white noise" models can be dangerous. Ecology Letters 2:228-232.

Northcote, T. G. 1997. Potamodromy in Salmonidae: living and moving in the fast lane. North American Journal of Fisheries Management 17:1029–1045.

Poff, N. L. 2002. Ecological response to and management of increased flooding due to climate change. Proceedings of the Royal Society of London (A) 360:1497-1510.

Rieman, B. E., and J. D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. General Technical Report INT-302. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Boise, Idaho.

Rieman, B., and J. Clayton. 1997. Wildfire and native fish: issues of forest health and conservation of sensitive species. Fisheries 22:6-15.

Sabo, J. L., and D. M. Post. 2008. Quantifying periodic, stochastic and catastrophic environmental variation. Ecological Monographs 78:19-40.

Seegrist, D. W., and R. Gard. 1972. Effects of floods on trout in Sagehen Creek, California. Transactions of the American Fisheries Society 101:478–482.

Schmetterling, D. A. 2001. Seasonal movements of fluvial westslope cutthroat trout in the Blackfoot River drainage, Montana. North American Journal of Fisheries Management 21:507–520.

Schmetterling, D. A. 2003. Reconnecting a fragmented river: movements of westslope cutthroat trout and bull trout after transport upstream of Milltown Dam, Montana. North American Journal of Fisheries Management 23:721–731.

Steele, J. H. 1985. A comparison of terrestrial and marine ecological systems. Nature 313:355-358.

Thurow, R. F., C. E. Corsi, and V. K. Moore. 1988. Status, ecology, and management of Yellowstone cutthroat trout in Upper Snake River Drainage, Idaho. American Fisheries Society Symposium 4:25-36.

Van Kirk, R. W., and S. L. Hill. 2007. Demographic model predicts trout population response to selenium based on individual-level toxicity. Ecological Modeling 206:407–420.

Vasseur, D. A., and P. Yodiz. 2004. The color of environmental noise. Ecology 85:1146-1152.

Waples R. S., T. Beechie, and G. R. Pess. 2008. Evolutionary history of Pacific salmon in dynamic environments. Evolutionary Applications 1:189-206.

Waples R. S., T. Beechie, and G. R. Pess. 2009. Evolutionary history, habitat disturbance regimes, and anthropogenic changes: what do these mean for resilience of Pacific salmon populations? Ecology and Society 14(1):3.

Watry, C. B. and D. L. Scarnecchia. 2008. Adfluvial and fluvial life history variations and migratory patterns of a relict charr *Salvelinus confluentus* stock in west-central Idaho, USA. Ecology of Freshwater Fish 17:231-243.

Westerling, A. L., H. Hidalgo, D. R. Cayan, and T. Swetnam. 2006. Warming and earlier spring increases western U.S. forest wildfire activity. Science 313:940–943.

Wigley T. M. L., R. L. Smith, and B. D. Santer. 1998. Anthropogenic influence on the autocorrelation structure of hemispheric-mean temperatures. Science 282:1676-1679.

Williams, J. E., A. L. Haak, H. Neville, and W. T. Colyer. 2009. Potential consequences of climate change to persistence of cutthroat trout populations. North American Journal of Fisheries Management 29:533-548.

Wilmers, C. C., E. Post, and A. Hastings. 2007. A perfect storm: the combined effects on population fluctuations of auto-correlated environmental noise, age structure, and density dependence. American Naturalist 169: 673-683.

Young, M. K, editor. 1995. Conservation assessment for inland cutthroat trout. General Technical Report RM-GTR-256. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Ft. Collins, CO.

APPROACHES IN IDENTIFYING POST-WILDFIRE POTENTIAL DEBRIS FLOWS AND REFUGE IN HEADWATER STREAMS

Ted R. Sedell

Department of Ecology, Montana State University, PO Box 173460, Bozeman, Montana 59717, 406-994-6643, tsedell@gmail.com

Robert E. Gresswell

USGS Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, Montana 59715, 406-994-7085, bgresswell@usgs.gov

Extended Abstract

Native trout populations in western North America have declined over the last century because of habitat fragmentation and degradation, and invasive species introductions (Behnke 1992). Negative effects associated with predicted climate change over the next century include increased water temperature, an altered precipitation pattern, and increased risk of wildland fire, and therefore, persistence of native trout populations is uncertain, particularly in isolated headwater streams (Williams et al. 2009). For example, with increased occurrences of wildfires, headwater streams may be especially susceptible to debris-flow torrents. Furthermore, because the probability of debris flow increases in landscapes that have recently burned, identifying those areas prior to the occurrence of wildfire could provide managers information necessary to plan management actions for protecting remnant fish populations in headwater streams.

Predicting the timing, extent, and severity of future wildfires or subsequent precipitation and runoff events is difficult because of the complex array of variables that influence these events responses, including annual variation of weather patterns. However, it is possible to identify channels within stream networks that may be prone to debris flows in a hypothetical post-wildfire event. To that end, we conducted both coarse- and fine-scale spatial analyses of debris flow potential and the spatial distribution and abundance of cutthroat trout *Oncorhynchus clarkii* in five high elevation headwater streams of the Colorado Rocky Mountains. Predictive models for debris-flow hazards can be applied before the occurrence of wildfires (with a projected burn severity distribution) to identify potentially susceptible drainage basins. Subsequently, results can be used to direct planning strategies that minimize the potential for catastrophic fires in those areas and identify fish populations at risk of extirpation. Our objectives were to (1) use channel and network morphology to predict potential locations of debris flows, and (2) estimate the overall risk of extirpation and availability of refugia for cutthroat trout.

Initially, we evaluated the likelihood that a debris-flow will occur in a post-wildfire environment and the channel conditions that lead to propagating debris flows and onset of deposition. Next, we assessed how much of stream channels would be affected by debris flows. We identified stream reaches that have an increased risk of post-wildfire debris flows using reach-level characteristics that include channel gradient and constraint, adjacent hillside slope, number of tributaries, and the confluence angle at which tributaries intersect the main channel. Debris flow probability was then estimated for each tributary using characteristic storm and burn scenarios, and geographic information describing topographic, soil, and vegetation variables. Channel reaches were mapped in five stream networks of the first and second order in the headwaters of the Colorado River to assess the potential for catastrophic pulse disturbance.

To assess the potential for debris flows, we used five major reach-scale characteristics that influence either the propagation of debris flows or the onset of deposition. These characteristics include channel gradient, channel constraints, hillslope angle, and confluence angle. Debris flows are likely to originate in tributaries and propagate into and along the main stem channel. Typically, destructive debris flows are found in systems that have steep, constrained channel gradients, and adjacent to hillslopes > 30 (Hungr et al. 2005; Gartner et al. 2008). Likewise, tributary debris flows are more likely to propagate into the main stem where the tributary confluence angle is < 65 (May and Gresswell 2004). Deposition begins where channel gradient is < 7%, the valley floor becomes unconstrained (valley width index - VWI > 2.5 active channel widths across), and the adjacent hillslope is < 30 (Hungr et al. 2005; Gartner et al. 2008). This information was summarized and incorporated into debris flow models; each characteristic was assigned a (+) if the feature supported propagation of the debris flow or a (-) if the feature supported the onset of deposition (Table 1).

To evaluate the probability that an individual tributary would produce a post-wildfire debris flow, we used a geographic information system model developed by Cannon et al. (2010) that predicts the probability of a debris flow event using topography, soil characteristics, and hypothetical burn and storm scenarios. In this study, we used a worst-case scenario of a high-severity wildfire covering 100% of the watershed area and 100-year storm event with an average rainfall of 54 mm in 2 h. If more than two tributaries enter into a reach, the highest debris flow probability value was used to illustrate the worst-case scenario.

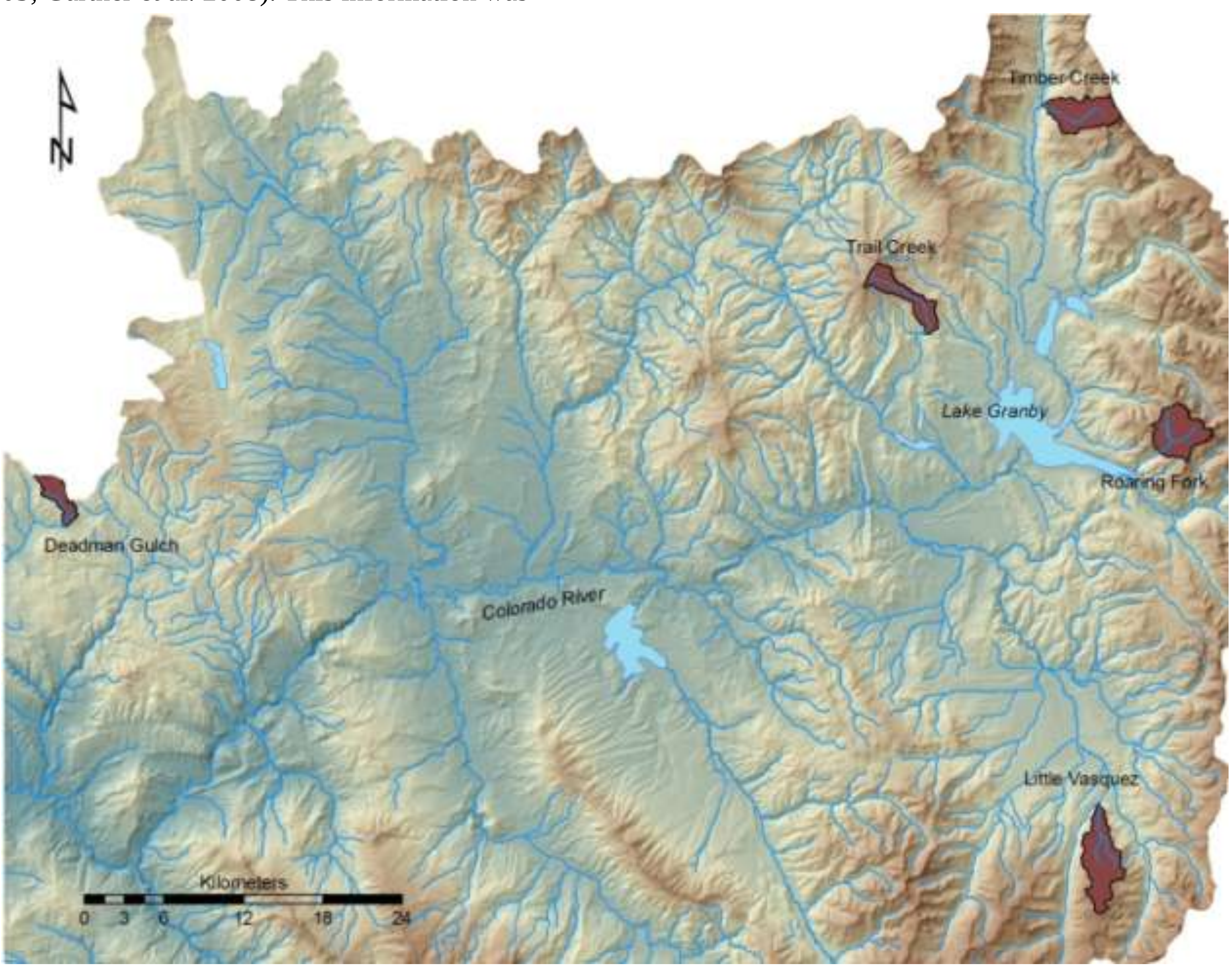


Figure 1. Location of the five watersheds (shown in dark red) in the upper Colorado River basin that were sampled during the summer low-discharge period (July-August), 2008-2009.

Table 1. Reach and tributary characteristics that influence post-wildfire debris flows in five watersheds sampled in the upper Colorado River basin, 2008-2009. A (+) represents a positive contribution to the propagation of a debris flow, (-) does not contribute to debris flow but potentially enhances onset of deposition, and (±) has no influence or is indeterminable on debris flow propagation or onset of deposition.

Watershed name	Reach number	Number of tributaries	Tributary debris flow probability (%)	Tributary confluence angle (< 65°)	Valley width index (< 2.5)	Channel gradient (> 7%)	Hillslope gradient (> 30%)
Deadman Gulch	1	0	–	–	(-)	(-)	(+)
	2	1	20	(+)	(-)	(-)	(-)
	3	1	9	(+)	(+)	(-)	(±)
	4	1	6	(+)	(-)	(-)	(±)
Little Vasquez	1	2	36	(+)	(+)	(+)	(±)
	2	0	–	–	(+)	(-)	(±)
	3	0	–	–	(+)	(-)	(-)
Roaring Fork	1	0	–	–	(+)	(+)	(+)
	2	1	< 1	(-)	(-)	(-)	(+)
	3	0	–	–	(+)	(+)	(±)
	4	0	–	–	(+)	(+)	(±)
	5	1	< 1	(-)	(-)	(-)	(±)
	6	0	–	–	(+)	(+)	(±)
Timber Creek	1	0	–	–	(-)	(-)	(-)
	2	0	–	–	(+)	(+)	(+)
	3	1	35	(-)	(+)	(+)	(±)
	4	2	15	(-)	(+)	(+)	(±)
	5	1	4	(+)	(+)	(+)	(+)
	6	0	–	–	(-)	(-)	(+)
Trail Creek	1	0	–	–	(+)	(-)	(±)
	2	0	–	–	(+)	(+)	(±)
	3	0	–	–	(-)	(-)	(+)
	4	0	–	–	(+)	(-)	(+)
	5	0	–	–	(+)	(+)	(-)

Of the five watersheds we evaluated in the upper Colorado River basin, three had tributaries with a debris flow probability > 20 % (Table 1). The risk of severe debris flows was limited spatially, and it was generally greatest in lower reaches of the watershed. The combination of a wide spatial distribution of fish and the limited spatial distribution and low probability of debris flow may increase the overall resiliency of the population to pulse disturbances in a post-wildfire environment. Ultimately, these data can be used to develop more comprehensive management strategies for restoration, protection, and post-disturbance remediation of headwater stream networks that support remnant populations of native fishes.

References

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society, Bethesda, Maryland.

Cannon, S. H., J. E. Gartner,M. G. Rupert,J. A. Michael, A. H. Rea, and C. Parrett. 2010. Predicting the probability and volume of postwildfire debris flows in the intermountain western United States. Geologic Society of America Bulletin 122(1-2): 127-144.

Gartner, J. E., S. H. Cannon, P. M. Santi, and V. G. Dewolfe. 2008. Empirical models to predict the volumes of debris flows generated by recently burned basins in the western U.S. Geomorphology 96: 339 -354.

Hungr, O., S. McDougall, and M. Bovis. 2005. Entrainment of material by debris flows. Pages 135-158 *in* M. Jakob and O. Hungr, editors. Debris-flow hazards and related phenomena. Praxis-Springer, Berlin Heidelberg.

May, C. L., and R. E. Gresswell. 2004. Spatial and temporal patterns of debris-flow deposition in the Oregon Coast Range, USA. Geomorphology 57: 135-149.

Williams, J. E., A. L. Haak, H. M. Neville, and W. T. Colyer. 2009. Potential consequences of climate change to persistence of cutthroat trout populations. North American Journal of Fisheries Man-Management 29: 533-548.

INTERSPECIES SYNCHRONY IN SALMONID DENSITIES AND LARGE-SCALE ENVIRONMENTAL CONDITIONS IN CENTRAL IDAHO

Timothy Copeland and Kevin A. Meyer
Idaho Department of Fish and Game, 1414 East Locust Lane, Nampa, ID 83709, Tel 1-208-465-8404; FAX 1-208-465-8434; tim.copeland@idfg.idaho.gov and kevin.meyer@idfg.idaho.gov

ABSTRACT — Abundance of fluvial salmonids varies substantially through time, often due to environmental stochasticity. The extent to which several sympatric species respond coherently to large-scale factors has been investigated rarely in freshwater fishes. We compared correlations among six salmonids at regional and site-specific levels and examined relationships between large-scale environmental variation and changes in salmonid density. Salmonid densities in central Idaho declined from the mid 1980s to the mid 1990s, and then rebounded through 2003. Most correlations in fish density among native salmonids were statistically positive at both regional and site-specific levels. In contrast, nonnative brook trout *Salvelinus fontinalis* were positively correlated to native salmonids at the regional scale but at not the site-specific scale, suggesting that synchrony between native and nonnative species was disrupted at small scales. Streamflow and the number of Chinook salmon *Oncorhynchus tshawytscha* redds (a surrogate for nutrient influx and therefore increased productivity to streams) were correlated with fish densities across the landscape. The importance of environmental conditions differed by species. Our finding that large-scale bioclimatic conditions influence the abundance of several salmonid species is important for fish managers charged with managing entire ecosystems with complex and sometimes sensitive species assemblages.

INTRODUCTION

Many empirical studies have demonstrated links between environmental variation and the abundance of salmonids in streams (e.g., Bradford 1999; Mote et al. 2003). At the largest scale, population characteristics of fluvial fishes are generally influenced by streamflow, the biological productivity of the system, and thermal regime. However, the interrelationship between fish and their environment is often complex. As an example, anadromous fishes are affected directly by ocean conditions such as upwelling (Scheuerell and Williams 2005) and indirectly because ocean conditions affect the inland environment via weather patterns (Cayan and Peterson 1989). Analyses of the relationships between fish and their environment at large scales provide an opportunity to investigate bioclimatic conditions that might produce correlated abundance patterns (McElhany et al. 2000), yet few studies to date have included multiple species of fluvial fishes.

In this study, we explore the synchrony among six salmonid species in central Idaho and examine likely relationships between salmonid abundance and environmental variables. The large number of observations in our study gave us considerable analytical power to distinguish regional effects, and provided us the opportunity to examine the data at more than one scale. Our objectives were (1) to compare synchrony in the densities of salmonids among spatial scales, (2) to explore the covariations of selected environmental variables to salmonid densities, and (3) to look for commonalities among species. We chose bioclimatic indices that we knew were influential, rather than attempting to include all potential variables.

METHODS

Study Area

The study area consisted of a large contiguous network of stream habitats in the Salmon River and Clearwater River watersheds, excluding the North Fork of the Clearwater River, which is isolated by Dworshak Dam (Figure 1). Land cover is chiefly composed of coniferous forests at higher elevations and sagebrush-grass steppe at lower elevations.

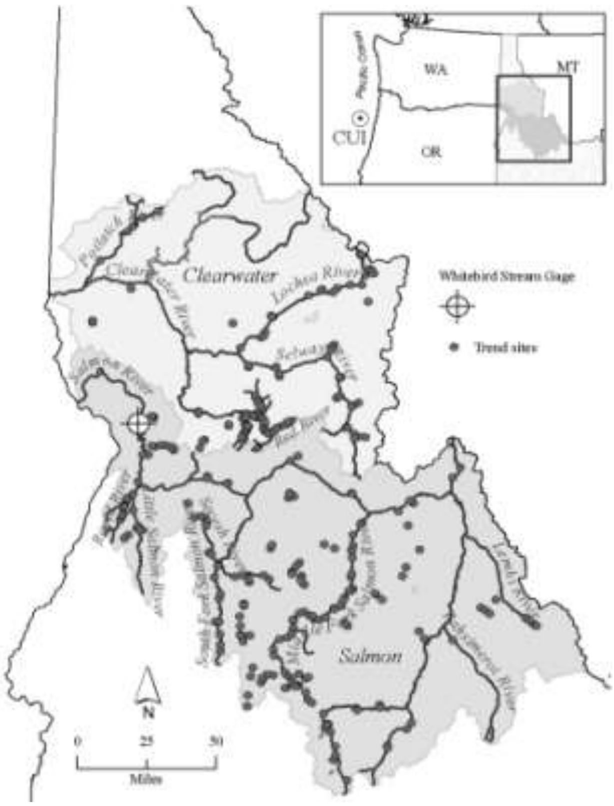


Figure 1. Distribution of 367 survey sites within the Salmon and Clearwater basins in Idaho and the location of the Whitebird stream gauging station. The inset displays the position of the study area in the Pacific Northwest and the location from which the coastal upwelling index (CUI) estimate was derived.

Geology is largely granite and is part of the Idaho Batholith. Over 80% of the study area is publicly owned and nearly 25% is designated wilderness, with many large expanses functioning as *de facto* wilderness. Precipitation ranges from 40 to 200 cm annually, depending on elevation, and falls mostly as snow during winter months.

Fluvial native salmonids inhabiting the study area include Chinook salmon *Oncorhynchus tshawytscha*, redband steelhead trout *O. mykiss gairdneri*, bull trout *Salvelinus confluentus*, westslope cutthroat trout *O. clarkii lewisi*, and mountain whitefish *Prosopium williamsoni*. Brook trout *S. fontinalis* were first introduced in Idaho in the late 1800s, and although the stocking of this species has ceased, self-sustaining populations now exist in many parts of the study area.

Fish Sampling

Since 1985, the Idaho Department of Fish and Game has conducted a large-scale snorkel survey program in central Idaho. Although originally designed to track trends for anadromous species, observations on all resident salmonids have been recorded as well. For logistical reasons, site selection was not random but based on professional judgment regarding habitat quality for juvenile anadromous salmonids and accessibility. Sites averaged 10.6 m in width (range 1.7 – 50.0 m). Petrosky and Holubetz (1986) provide a more detailed description of sampling site selection. Because many sites were not surveyed consistently, we set a criterion of using sites where multiple (i.e., >3) sample events occurred in each decade (1980s, 1990s and 2000s). This resulted in a sample of 367 sites with 4,752 snorkel surveys.

Counts, expressed as number/100 m^2, were used as an index of abundance for each species. Depending on the stream width and water clarity, from one to five observers snorkeled slowly upstream counting all salmonids. Although rarely encountered, hatchery rainbow trout were distinguishable from wild *O. mykiss* by size and fin condition, and were excluded from our analyses. Visibility was recorded before each survey and averaged 4.3 m; 95% of surveys were conducted when visibility was ≥1.7 m. Water temperature during snorkel surveys averaged 14.5°C. Total counts were used as minimum abundance estimates with no correction for probability of detection, and thus should not be viewed as accurate estimates of true abundance. However, we assumed that this technique would function adequately as an index of abundance.

Our working hypothesis was that temporal changes in salmonid abundance at the landscape scale are influenced by large-scale bioclimatic variables, chiefly weather and marine nutrient inputs in the form of salmon. We chose five variables on which to focus: streamflow (FLOW), the Palmer drought severity index (PDSI), mean July air temperature (AirT), a coastal upwelling index (CUI), and counts of Chinook salmon redds (REDDS). We obtained estimates of the regressors from several sources. To represent unregulated streamflows in central Idaho, we obtained mean annual discharge (m^3/s) data on the lower Salmon River near Whitebird from the U.S. Geological Survey. The PDSI and average July air temperatures for central Idaho were computed by the National Climatic Data Center. The PDSI index is based on balance between moisture supply, soil characteristics, and evapotranspiration. The October CUI at 45°N 125°W was estimated by Scheuerell and Williams (2005) to characterize the ocean conditions that juvenile Chinook salmon from the Snake River encounter during their first season in the ocean. Lastly, we obtained redd counts at trend transects across central Idaho (www.streamnet.org).

To refine our working hypothesis into a mechanistic explanation for observed fish densities, we correlated each variable against species densities at lags from 0 to 5 years. We assumed the environment would not necessarily affect a species in the year in which the environmental variable was measured but should have a lag effect based on processes such as recruitment. Year 0 was not included for REDDS and CUI because those variables were measured after snorkel surveys were completed. Patterns in the lagged correlations were compared among species and the final form for each variable was chosen based on the largest number of potentially important correlations (r^2>0.10) in the same direction across the six species (data not shown). We chose to average FLOW at 3- and 4-year lags and PDSI at 4- and 5-year lags. We chose the 2-year lag for CUI and a 1-year lag for REDDS. Correlations were maximized for AirT with no lag.

We used correlation analyses to examine relationships among fish species. All fish densities were log-transformed ($\log_e$[density+0.01]) before analyses. At the regional scale (Clearwater and Salmon basins), we calculated the annual mean density of each species, then assessed the level of correlation among all species over time. At the site-level scale, we analyzed the relationships between species by correlation, using each snorkeling event as an observation and disregarding the year of sampling. We excluded pair-wise comparisons where density was zero for both species. Correlations were considered statistically significant at = 0.05. We conducted separate analyses for the Salmon and Clearwater drainages to discern whether the correlations were consistent between drainages.

To assess the relationship between the five environmental variables and the densities of the six salmonid species, we used canonical correlation. The canonical correlation was performed on the set of transformed densities averaged within year versus the regressors. We used the variable loadings and redundancies on the first canonical axis, as well as canonical cross loadings, to interpret the canonical correlation. Because correlation matrices were used, interpretation in terms of species density was made in relative terms and each species had equal weight. To clarify interpretation of the canonical correlation, we used information-theoretic model selection techniques to examine the univariate effects of the regressors on individual species densities (Burnham and Anderson 2002).

Univariate model selection for each species was based on Akaike's information criterion corrected for small sample sizes (Burnham and Anderson 2002). All additive combinations of the regressors were fit against transformed density for each species. As a general index of model fit, we used adjusted R . The relative likelihood of each model in the set of models for each species was assessed with Akaike weights (Burnham and Anderson 2002). The weight of evidence for a single regressor is evaluated from the Akaike weight of that regressor versus the others. The relative importance of a particular regressor, given all potential additive combinations of the regressors, was computed as the sum of the Akaike weights of all models containing that regressor (importance value).

Results

Average densities of all species declined from the 1980s to the mid 1990s, and then rebounded through 2003, although there was much variation about the trends. Bull trout and brook trout had the lowest average densities during the study period (Figure 2). The highest densities were observed for steelhead trout and Chinook salmon. Of all the salmonids, Chinook salmon had the most variable density and the most exaggerated trend. Densities of westslope cutthroat trout and mountain whitefish were intermediate to the other species. Average salmonid densities over time were strongly correlated among species at the regional scale, with 25 of the 30 correlation coefficients for each pair-wise comparison being positive and 13 being statistically significant at = 0.05 (Table 1).

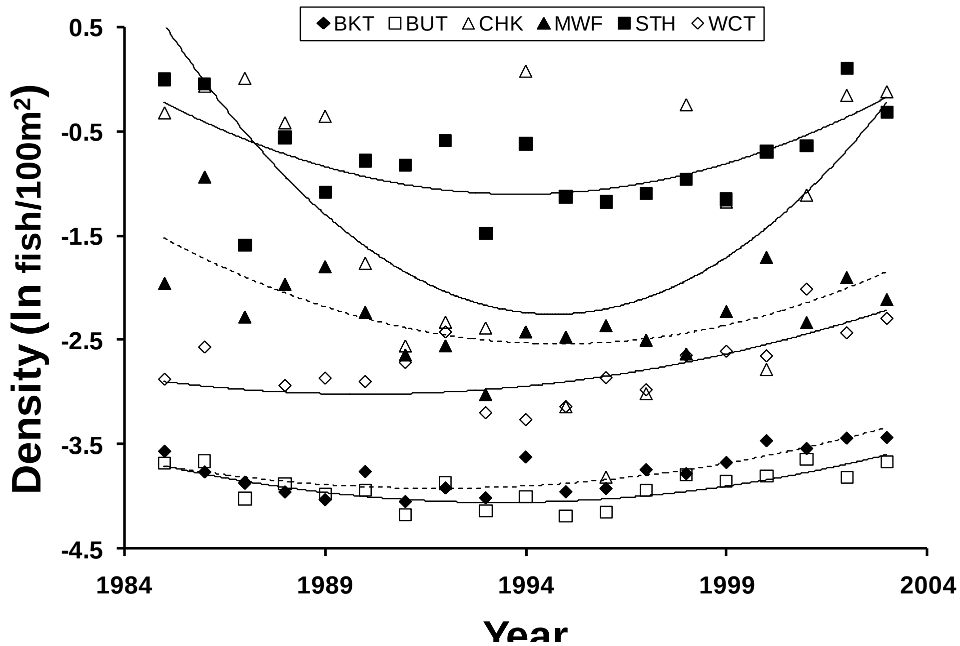


Figure 2. Density of six salmonid species in central Idaho, 1985-2003. Species were steelhead (STH), Chinook salmon (CHK), mountain whitefish (MWF), westslope cutthroat trout (WCT), brook trout (BKT), and bull trout (BUT). Trend lines were fitted using a second-order polynomial.

Table 1. Correlation coefficients among species for annual mean density at snorkel sites in the Salmon River (below the diagonal, *n* = 206) and Clearwater River (above the diagonal, *n* = 161) drainages. Species are brook trout (BKT), bull trout (BUT), Chinook salmon (CHK), mountain whitefish (MWF), steelhead trout (STH), and westslope cutthroat trout (WCT).

Species	BKT	BUT	CHK	MWF	STH	WCT
BKT	--	-0.07	-0.08	0.35	0.33	0.39
BUT	0.81*	--	0.51*	0.63*	0.45	0.52*
CHK	0.35	0.41	--	0.22	-0.05	0.04
MWF	-0.01	-0.01	0.41	--	0.67*	0.54*
STH	0.58*	0.65*	0.61*	0.48*	--	0.66*
WCT	0.23	0.36	0.31	0.54*	0.58*	--

$P < 0.05$

At the site-level scale, correlations among species were also frequent. In fact, 17 of the 30 pairwise comparisons had statistically significant correlations, 14 of which were positive (Table 2). Considering only native salmonids, 16 of the 20 comparisons were positively correlated at a statistically significant level, whereas only 2 statistically significant correlations were negative. However, of the 10 comparisons of nonnative brook trout density to that of native salmonids, only one relationship was statistically significant (brook trout and steelhead trout in the Salmon sub-basin, a negative correlation). In contrast, brook trout were positively correlated for 7 of 10 comparisons with native salmonids at the regional scale, 2 of which were statistically significant (Table 1).

Canonical Correlation

The canonical correlation explained a significant amount of the shared multivariate variance between fish densities and environmental variables (Wilks' $\Lambda = 0.05$, $df_1 = 30$, $df_2 = 34$, $P = 0.0006$). The redundancy of the original densities with the first axis of canonical density variate was 51.6% and declined to <10% for the other axes, so we interpreted only the first axis.

Table 2. Correlation coefficients among species for all pair-wise comparisons from each snorkeling event (*n* = 4,752) in the Salmon River (below the diagonal) and Clearwater River (above the diagonal) drainages. Sample size for each comparison is in parentheses. See Table 1 for species codes.

Species	BKT	BUT	CHK	MWF	STH	WCT
BKT	--	-0.20 (40)	-0.09 (206)	0.01 (229)	-0.04 (209)	0.04 (167)
BUT	0.05 (161)	--	0.66* (207)	0.50* (208)	0.18* (303)	-0.02 (264)
CHK	0.07 (467)	0.10* (423)	--	0.68* (702)	0.15* (790)	0.17* (628)
MWF	0.00 (296)	0.14* (407)	0.21* (1039)	--	0.28* (852)	0.19* (715)
STH	-0.12* (497)	0.06 (637)	-0.11* (1307)	0.11* (1293)	--	0.08* (961)
WCT	0.04 (137)	0.09 (290)	0.01 (564)	0.15* (738)	-0.10* (791)	--

$^*P < 0.05$

Table 3. Contributions of individual variables to the canonical correlation between fish densities and selected environmental variables.

Variable	First axis loading	Canonical cross loading
	Fish densities	
BKT	0.735	0.703
BUT	0.848	0.812
CHK	0.766	0.732
MWF	0.664	0.635
STH	0.721	0.689
WCT	0.538	0.515
	Regressors	
AirT	0.508	0.486
CUI	-0.608	-0.581
Flow	0.684	0.654
PDSI	0.612	0.586
Redds	0.708	0.678

Contributions to the canonical correlation differed among fish species and environmental variables. All species densities loaded positively onto the first axis of the dependent canonical variate, bull trout most strongly, then Chinook salmon, brook trout, steelhead trout, mountain whitefish, and westslope cutthroat trout, respectively (Table 3). Order by canonical cross-loadings was the same. Of the environmental variables, REDDS loaded most strongly on the first axis, followed by FLOW, CUI, PDSI, and AirT, respectively (Table 3). As with densities, the order by canonical cross loadings was the same. All the regressors loaded positively on the first axis except for CUI.

Univariate Model Selection

Akaike weights showed that importance of the regressor variables differed among species (Table 4). For bull trout and brook trout density, FLOW and AirT were the two most important regressor variables (positive effects). For mountain whitefish, CUI was the most important regressor followed by PDSI (both negative relationships). Coastal upwelling index was also the most important regressor for steelhead trout density (negative relationship), but secondarily, AirT was an important regressor (positive relationship). Not surprisingly, REDDS was by far the most important regressor for Chinook salmon density (positive relationship). For westslope cutthroat trout, there was much uncertainty in model selection because no model explained a large amount of variance for this species (maximum adjusted R = 0.31). More variance was explained for the other species, with maximum adjusted R ranging from 0.52 for Chinook salmon to 0.69 for mountain whitefish.

Discussion

Our results suggest that large-scale temporal trends in abundance were consistent across all study species in central Idaho from the mid 1980s to mid 2000s. The study species were all fluvial salmonids that arguably should react similarly to large-scale environmental variation. However, these species have disparate thermal optima, spawn and migrate at various times of the year in dissimilar habitats and conditions, prefer different local micro- and meso-habitats for rearing, and make use of resident and anadromous life history behaviors. Consistent trends in abundance amid such diverse phenotypic characteristics suggest that regionally coherent bioclimatic controls were likely driving the synchronized relationships.

Table 4. Akaike weights and importance values of regressors explaining observed fish densities by species.

Variable	Akaike weight	Importance value
	Brook trout	
AirT	0.018	0.845
CUI	0.002	0.134
FLOW	0.037	0.944
PDSI	0.004	0.537
REDDS	0.006	0.330
	Bull trout	
AirT	0.001	0.727
CUI	0.003	0.577
FLOW	0.078	0.982
PDSI	0.002	0.571
REDDS	<0.001	0.230
	Chinook salmon	
AirT	0.001	0.181
CUI	0.005	0.314
FLOW	0.003	0.375
PDSI	0.010	0.401
REDDS	0.098	0.949
	Mountain whitefish	
AirT	<0.001	0.182
CUI	0.024	0.946
FLOW	0.004	0.358
PDSI	0.026	0.727
REDDS	<0.001	0.141
	Steelhead trout	
AirT	0.009	0.932
CUI	0.022	0.958
FLOW	0.002	0.140
PDSI	0.002	0.199
REDDS	0.006	0.164
	Westslope cutthroat trout	
AirT	0.108	0.436
CUI	0.087	0.375
FLOW	0.068	0.413
PDSI	0.044	0.413
REDDS	0.102	0.339

In addition to environmental effects, other potentially synchronizing processes include dispersal and the effects of mobile predators or parasites (Liebhold et al. 2004). We examined interspecies synchrony, not populations *per se*; hence, dispersal cannot explain the patterns we observed (Liebhold et al. 2004). Human predation cannot explain the interspecies patterns in this study, because there was no fishery for wild steelhead or Chinook salmon during the study period, and because much of the study area is exceedingly rugged and remote with concomitant negligible exploitation rates. Whirling disease *Myxobolus cerebralis* is another potential means of synchronized mortality, but it does not affect all salmonids equally, it is not found throughout the study area, and resistance to it is unlikely to develop quickly enough to explain the increased densities observed after the mid 1990s. By elimination, we concluded that the most likely synchronizing process in our study was shared correlation to environmental conditions (i.e., the Moran effect, Moran 1953).

Bioclimatic Effects

Most of the environmental variables included in our analyses logically should have affected salmonid density in subsequent years. For example, CUI was measured in October, at least a month after surveys were conducted in a given year. Only AirT was most strongly related to salmonid densities with no lag. For all other environmental variables, lags of between one and five years were most strongly related to salmonid density, presumably by impacting recruitment (e.g., Tedesco et al. 2004; Zorn and Nuhfer 2007a). We detected these effects in subsequent years through changes in fish densities.

Considering the salmonid community as a multivariate whole, FLOW and REDDS were the most important environmental variables and AirT was the least important. Flow effects on salmonid recruitment (via juvenile survival) are well-documented and can be seasonal in nature, such as redd scour during incubation (Carline and McCullough 2003), displacement of newly emerged fry (Zorn and Nuhfer 2007b), and reduced overwinter survival during low flow years (Mitro et al. 2003). Crozier and Zabel (2006) suggested annual stream flow (dictated by winter/spring snowpack) affected parr-to-smolt survival of Chinook salmon in the Salmon River basin. Hydrographs of almost all streams in central Idaho are driven by snowmelt with flows peaking in May or June. Given this similarity in hydrologic regime, it is plausible that flow would affect stream fishes simultaneously over a large area. In our study, FLOW was of primary importance for bull trout and brook trout, but its effect was positive for all species.

In the multivariate model, REDDS was important largely because of the high loading of Chinook salmon into the canonical variate. It was not surprising that a one-year lag on REDDS was the most influential environmental variable for Chinook salmon because most Chinook (>97%) observed were age 0. We hypothesized that redd counts also might be influential for other species because of the influence of marine-derived nutrients on stream productivity (Wipfli et al. 1998). The fact that for all species the strongest lag for REDDS was one year suggests that increased numbers of decaying adults led to higher survival (presumably through increased stream productivity) to the next year for all species. However, in the univariate models, REDDS had a minor effect for all species except Chinook salmon, suggesting this influence was weak, at least given the current low abundances of adult Chinook salmon in the system.

Univariate models suggested that CUI was important to steelhead and mountain whitefish. The form of CUI we used was measured in October and was chosen because of its demonstrated impact on Chinook salmon (Scheuerell and Williams 2005). It may have acted as a surrogate for fall stream conditions via influence of ocean conditions on inland weather patterns (Cayan and Peterson 1989). Fall is when steelhead adults are migrating up the Columbia and Snake rivers to Idaho (August-October) and when mountain whitefish spawn. Because CUI was most strongly related to steelhead and whitefish at a lag of four years, it is likely that CUI was related to recruitment of these species, not adult density directly. Alternatively, the relationship between CUI and steelhead or whitefish may have been non-causative. An apparent environmental effect may arise because two species that are correlated to each other such that the first species shows an apparent relationship to that particular environmental variable, even though the direct effect is actually on the second species. There may be other plausible indirect effects as well, providing a good illustration of why it is hard to infer causal mechanisms with large-scale data.

Temperature in the year of observation was an important modifier for all species except Chinook salmon and mountain whitefish. Because there was no lag time for the effect of AirT, we hypothesized this relationship was related to detection probability, not density. Temperature is related directly to the probability of detecting bull trout, cutthroat trout, and rainbow trout with snorkel surveys because fish become less active and more difficult to observe at low temperatures (Thurow et al. 2006). Chinook salmon and mountain whitefish are schooling species, the detection of which may be less affected by temperature because it is easier to observe schools of fish than dispersed individuals. However, temperature was not of primary importance for any species.

The relationship between environmental conditions and fish density was weakest for westslope cutthroat trout. On average, correlations to environmental variables for this species were less than half the values for other species. Model selection results were ambiguous for cutthroat trout; all importance values of the regressors were <0.50. Further, this species had the lowest loading in the canonical correlation. Westslope cutthroat trout are closely associated with headwater habitats, which are typically more stochastic than downstream reaches and therefore may be less likely to be influenced by the large-scale environmental variables we analyzed.

In spatiotemporal population dynamics, there are two components of variation: environmental correlations and a demographic component, i.e., density-dependence (Liu et al. 2009). In our analyses, the density-dependent component was incorporated only for Chinook salmon (via REDDS), and it was very significant in multivariate and univariate models, as has been found previously in Chinook salmon populations in the study area (Copeland et al. 2008). Similar indices of cohort egg density were not available or were incomplete for the other species, but their inclusion likely would explain more variance in salmonid density.

Synchrony Among Salmonid Species

High et al. (2008) found that population abundance in salmonid communities in central Idaho fluctuated in synchrony over a multi-decadal time scale, and our study demonstrates that this pattern carries through to smaller scales. Synchrony is common across animal populations and should be expected in multi-population studies (Liebhold et al. 2004; Bjørnstad et al. 2009), including studies of fish communities (Wood and Austin 2009). However, the data needed to discern this phenomenon are extensive in both time and space, and consequently are scarce in freshwater fish ecology. Most previous studies of population synchrony in freshwater fishes have examined only one or two species (e.g., Isaak et al. 2003; Zorn and Nuhfer 2007a). In one of the few studies of synchrony in multiple freshwater fish species, Tedesco et al. (2004) found synchrony in 4 species of African fishes was influenced by regional river discharge the previous year over a 24-year time frame. In comparison, we found synchrony among 6 species over 19 years.

Niche partitioning may allow native salmonids in central Idaho to live sympatrically and fluctuate communally, as demonstrated by synchrony at both the regional and site scales. Such niche partitioning is common among sympatric native salmonids (e.g., Nakano et al. 1992). The presence of exotic salmonids can disrupt this partitioning (Hasagawa and Maekawa 2006) and may explain why the synchrony we observed between nonnative brook trout and native salmonids at the regional scale was absent at the site scale. Indeed, negative interactions between brook trout and native salmonids are ubiquitous in western North America (e.g., Levin et al. 2002; Rieman et al. 2006).

For fishery managers, interspecies synchrony may allow a more powerful multispecies approach to monitoring fish populations. Regional synchrony allows establishment of trend sites that capture the ecological variation over a larger area. By using targeted surveys to describe temporal trends, managers can better interpret findings from individual surveys within a region and be more informed about the effects of management actions (Zorn and Nuhfer 2007a). However, the extent of synchrony should be estimated, because it will vary depending on the study area, the focal group of species, the influential environmental variables, and the range of environmental conditions.

The occurrence of interspecies synchrony suggests that abundances of all salmonids in central Idaho currently are linked. All native species in our study area were formerly more abundant, and may have been less synchronous under such conditions (see Isaak et al. 2003 for an example using Chinook salmon). Synchrony theoretically reduces metapopulation resilience (Heino et al. 1997), and this effect may carry through to community structure.

Studies such as ours serve as an important baseline from which to understand the effects of climate change on stream fishes. There is a mismatch of scales between aquatic ecology and climatology; climate studies are conducted at very large scales (e.g., continental), whereas the typical scale of ecological studies is much smaller. In our data, the clearest climatic influence on fish density was the direct correlation between FLOW and bull trout density, a species that has already been used as an indicator of climate change effects (Rieman et al. 2007). These authors found that bull trout populations in Idaho were sensitive to temperature, but our results suggest that changes to the flow regime also should be investigated. Effects of various climatic variables on fish populations tend to be interdependent and mechanistically complex. Further, the synchronizing agent may be different from causes responsible for oscillations (Moran 1953; Bjørnstad et al. 2009). However, this study provides an inductive basis for further exploration of likely mechanisms influencing the dynamics and structure of fluvial salmonid communities.

The bioclimatic conditions we included explained 52% of the variation in the abundance of salmonids in central Idaho. Clearly, other factors also were affecting the populations in our study. The relative contribution of the bioclimatic variables to overall variation may provide managers a relative sense of what variation in abundance may be under management control. Such knowledge at large scales may help focus management efforts or shape restoration actions. The influence of climate on fishes is complex and can vary across differing scales in space and time. Causal mechanisms are difficult to infer and require further research. Our study provides a basis for suggesting fruitful avenues for such work.

Acknowledgments

Bonneville Power Administration provided funding for this work (Project 1991-073-00). We are indebted to C. Petrosky and D. Isaak for providing comments on an earlier draft that greatly improved the manuscript, and Liz Mamer and Paul Bunn for compiling the fish dataset.

References

Bjørnstad, O. N., A. M. Liebhold, and D. M. Johnson. 2009. Transient synchronization following invasion: revisiting Moran's model and a case study. Population Ecology 50:379-389.

Bradford, M. J. 1999. Temporal and spatial trends in the abundance of coho salmon smolts from western North America. Transactions of the American Fisheries Society 128:840-846.

Burnham, K. P, and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach, 2nd edition. Springer-Verlag, New York.

Carline, R. F., and B. J. McCullough. 2003. Effects of floods on brook trout populations in the Monongahela National Forest, West Virginia. Transactions of the American Fisheries Society 132:1014-1020.

Cayan, D. R., and D. H. Peterson. 1989. The influence of North Pacific atmospheric circulation on streamflow in the West. Geophysical Monograph 55:375-397.

Copeland, T., J. Johnson, and P. R. Bunn. 2008. Idaho natural production monitoring and evaluation project, 2007 annual report. Idaho Department of Fish and Game Report 08-08. Boise, Idaho.

Crozier, L., and R. W. Zabel. 2006. Climate impacts at multiple scales: evidence for differential population responses in juvenile Chinook salmon. Journal of Animal Ecology 75:1100-1109.

Hasagawa, K., and K. Maekawa. 2006. The effects of introduced salmonids on two native stream-dwelling salmonids through interspecific competition. Journal of Fish Biology 68:1123-1132.

Heino, M., V. Kaitala, E. Ranta, and J. Lindström. 1997. Synchronous dynamics and rates of extinction in spatially structured populations. Proceedings of the Royal Society of London B 264:481-486.

High, B., K. A. Meyer, D. J. Schill, and E. R. J. Mamer. 2008. Distribution, abundance, and population trends of bull trout in Idaho. North American Journal of Fisheries Management 28:1687-1701.

Isaak, D. J., R. F. Thurow, B. E. Rieman, and J. B. Dunham. 2003. Temporal variation in synchrony among Chinook salmon (*Oncorhynchus tshawytscha*) red counts from a wilderness area in central Idaho. Canadian Journal of Fisheries and Aquatic Sciences 60:840-848.

Levin, P. S., S. Achord, B. E. Feist, and R. W. Zabel. 2002. Non-indigenous brook trout and the demise of Pacific salmon: a forgotten threat? Proceedings of the Royal Society of London 269: 1663:1670.

Liebhold, A., W. D. Koenig, and O. Bjørnstad. 2004. Spatial synchrony in population dynamics. Annual Review of Ecology, Evolution, and Systematics 35:467-490.

Liu, Z., M. Gao, Z. Li, and G. Zhu. 2009. Synchrony of spatial populations: heterogeneous population dynamics and reddened environmental noise. Population Ecology 51:221-226.

McElhaney, P., M. H. Ruckleshaus, M. J. Ford, T. C. Wainright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. NOAA Technical Memorandum NMFS-NWFSC-42.

Mitro, M. G., A. V. Zale, and B. A. Rich. 2003. The relation between age-0 rainbow trout (*Oncorhyn-*

chus mykiss) abundance and winter discharge in a regulated river. Canadian Journal of Fisheries and Aquatic Sciences 60:135-139.

Moran, P. A. P. 1953. The statistical analysis of the Canadian lynx cycle. Australian Journal of Zoology 1:291-298.

Mote, P. W., E. A. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. W. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: the water, salmon, and forests of the Pacific Northwest. Climatic Change 61:45-88.

Nakano, S., K. D. Fausch, T. Furukawa-Tanaka, K. Maekawa, and H. Kawanabe. 1992. Resource utilization by bull char and cutthroat trout in a mountain stream in Montana, U.S.A. Ichthyological Research 39:211-217.

Petrosky, C. E., and T. C. Holubetz. 1986. Idaho habitat evaluation of off-site mitigation record. Annual report to Bonneville Power Administration, Project 83-7. Idaho Department of Fish and Game, Boise, Idaho.

Rieman, B. E., D. Isaak, S. Adams, D. Horan, D. Nagel, C., Luce, and D. Myers. 2007. Anticipated climate warming effects on bull trout habitats and populations across the Interior Columbia River Basin. Transactions of the American Fisheries Society. 136:1552-1565.

Rieman, B. E., J. T. Peterson, and D. L. Myers. 2006. Have brook trout (*Salvelinus fontinalis*) displaced bull trout (*Salvelinus confluentus*) along longitudinal gradients in central Idaho streams? Canadian Journal of Fisheries and Aquatic Sciences 63:63-78.

Scheuerell, M. D., and J. G. Williams. 2005. Forecasting climate-induced changes in the survival of Snake River spring/summer Chinook salmon (*Oncorhynchus tshawyscha*). Fisheries Oceanography 14:448-457.

Tedesco, P. A., B. Hugueny, D. Paugy, and Y. Fermon. 2004. Spatial synchrony in population dynamics of West African fishes: a demonstration of an intraspecific and Interspecific Moran effect. Journal of Animal Ecology 73:693-705.

Thurow, R. F., J. T. Peterson, and J. W. Guzevich. 2006. Utility and validation of day and night snorkel counts for estimating bull trout abundance in first- to third-order streams. North American Journal of Fisheries Management 26:217-232.

Wipfli, M. S., J. Hudson, and J. Caouette. 1998. Influence of salmon carcasses on stream productivity: response of biofilm and benthic macroinvertebrates in southeastern Alaska, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 55:1503-1511.

Wood, R. J., and H. M. Austin. 2009. Synchronous multidecadal fish recruitment patterns in Chesapeake Bay, USA. Canadian Journal of Fisheries and Aquatic Sciences 66:496-508.

Zorn, T. G., and A. J. Nuhfer. 2007a. Regional synchrony of brown trout and brook trout population dynamics among Michigan rivers. Transactions of the American Fisheries Society 136:706-717.

Zorn, T. G., and A. J. Nuhfer. 2007b. Influences on brown trout and brook trout population dynamics in a Michigan river. Transactions of the American Fisheries Society 136:691-705

Sensitivity and Vulnerability of Brook Trout Populations to Climate Change

Brad Trumbo and Mark Hudy
U.S. Forest Service, Fish and Aquatic Ecology Unit, James Madison University, Mail Stop Code 7801, Harrisonburg, Virginia 22807, USA

Eric P. Smith and Dong-Yun Kim
Department of Statistics, Virginia Polytechnic Institute, Blacksburg, Virginia 24061, USA

Bruce A. Wiggins
Department of Biology, James Madison University, mail Stop Code 7801, Harrisonburg, Virginia 22807, USA

Keith H. Nislow
Northern Research Station, U.S. Forest Service, Amherst, Massachusetts, 01003, USA

Charles A. Dolloff
Southern Research Station, U.S. Forest Service, Blacksburg, Virginia, 24061, USA

ABSTRACT — Predicting future brook trout *Salvelinus fontinalis* distributions at the population scale under various climate scenarios is of interest to the Eastern Brook Trout Joint Venture. Previous larger scale models have been useful in highlighting the potential threat; however, the predicted air and water temperature errors associated with these models makes predictions of the persistence of individual brook trout populations problematic. We directly measured paired air and water temperatures in watersheds (N = 77) containing reproducing populations of brook trout in Virginia. We found that paired air and water temperature relationships are highly variable among patches but are a useful dataset to classify sensitivity and vulnerability of existing brook trout patches. We developed a classification system using sensitivity and vulnerability metrics that classified sampled brook trout habitats into four categories (High Sensitivity- High Vulnerability (51.9%); High Sensitivity-Low Vulnerability (10.4%); Low Sensitivity-High Vulnerability (7.8%); Low Sensitivity-Low Vulnerability (29.9%). Our direct measurement approach identified potential refugia for brook trout at lower elevations and with higher air temperatures than previous larger scale modeling efforts. Our sensitivity and vulnerability groupings should be useful for managers making investment decisions in protecting and restoring brook trout.

Introduction

Although no known brook trout *Salvelinus fontinalis* populations have been extirpated because of climate change effects (Hudy et al. 2008), several studies have identified the potential threat of air temperature increases in dramatically reducing the current range of brook trout in the eastern United States (Meisner 1990; Flebbe 1994; Clark et al. 2001; Flebbe et al. 2006). While useful in highlighting the potential threat from increases in air temperature, the errors associated with models using secondary data (predicted air temperatures (PRISM 2007); and predicted water temperature response relative to predicted air temperature) makes predictions of the persistence of individual brook trout populations under various climate change scenarios problematic (Johnson 2003). Models using secondary data often ignore site-specific landscape characteristics that may influence the relationship between air and water temperatures. Predictions of habitat loss based on models that assume a simple positive direct relationship between air and water temperature across all habitats are likely to be overly pessimistic (Meisner 1990; Flebbe 1994; Keleher and Rahel 1996; Rahel et al. 1996; Clark et al. 2001;

Flebbe et al. 2006; Rieman et al. 2007; Williams et al. 2009). Some brook trout habitats may persist even under the most pessimistic climate change scenarios due to localized landscape conditions. Variability in the relationship between water temperature and air temperature can be quantified (Cluis 1972; Pilgrim et al. 1998; Mohseni and Stefan 1999; Isaak and Hubert 2001; Johnson 2003) and effectively used by managers to rank the resistance of individual brook trout populations to various climate change scenarios. Identifying brook trout habitats that are more resistant to air temperature increases is an important step in prioritizing the restoration and conservation work of the Eastern Brook Trout Joint Venture (EBTJV 2006). Our pilot studies and earlier research (Fink 2008) suggest that the relationship between air and water temperature is (1) highly variable at the current brook trout population scale and (2) influenced by local conditions and their interactions (i.e. elevation, aspect, topography shading, riparian cover, latitude, longitude, insolation and ground water sources). The influence of these characteristics at localized scales appears to play a more important role than expected in stream thermal stability (Meisner 1990; Pilgrim et al. 1998; Moore et al. 2005; Wehrly et al. 2007; Fink 2008).

The specific objectives of this study are to (1) quantify the variability in the daily maximum air and daily maximum water temperature responses during the water temperature stress period (July 1 to September 30) for brook trout populations in Virginia and (2) develop a classification system using sensitivity and vulnerability metrics that will be of use to managers in prioritizing their work for brook trout.

Methods

Study Area, Sample Unit Delineation and Selection

This project includes all habitats with reproducing populations of brook trout within the state of Virginia. Brook trout presence-absence data from the EBTJV (Mohn and Bugas 1980; EBTJV 2006; Hudy et al. 2008) were overlaid on catchments from the National Hydrography Plus (NHD+) dataset (USGS 2008) to produce a dataset of catchments containing reproducing populations of brook trout. Contiguous catchments containing brook trout were then dissolved into individual watersheds or “habitat patches” of reproducing brook trout. Each patch was presumed to be isolated (reproductively) from other patches. A total of 272 patches were found in Virginia. Candidate landscape metrics hypothesized to be important to potential air temperature and water temperature relationships were summarized in a Geographic Information System (GIS) for both the watershed area above the pour point, and the watershed area above the patch centroid (Table 1). The pour point is the intersection of the stream segment in the NHD+ dataset and the most downstream brook trout occupied catchment boundary. The centroid location of the brook trout habitat patch was determined by a GIS algorithm and then snapped to the nearest stream channel (Figure 1). A cluster analysis (Ward’s Method; SAS 2000) was used to group the 272 patches into 9 groups (see table 1 for grouping metrics). We then systematically selected

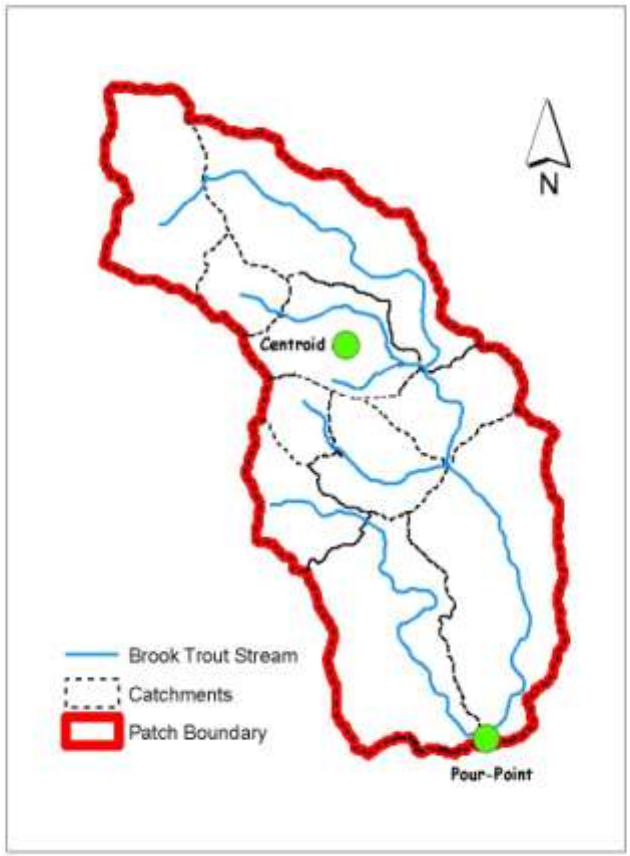


Figure 1. Contiguous catchments containing brook trout were dissolved into “patches” of reproducing brook trout habitat. Paired air and water temperature thermographs were placed at pour point and at the stream section nearest to the centroid to directly measure water temperature responses to air temperatures.

Table 1. Candidate landscape metrics summarized for each patch or watershed above each centroid (sample areas = patch or centroid watershed). Metrics followed by an asterisk (*) were used for the cluster analysis subsampling protocol.

Metric	Units	Source
Sample Area	ha	derived
Riparian Area sample area	ha	derived (100m buffer NHD+)
Total Annual Solar Gain sample area	kWh	Fu and Rich 1999
Total Annual Solar Gain in riparian sample area*	kWh	Fu and Rich 1999
Mean Solar Gain (July 1-September 300) in Riparian (100 m buffer) corrected for the percentage of canopy cover	kWh/30mpixel	Fu and Rich 1999
Mean Annual Solar Gain in riparian sample area	kWh/30mpixel	Fu and Rich 1999
Sample point Elevation*	m	derived
Sample point 30-year Mean Max Temp*	Celsius	PRISM 2007
% groundwater flow in sample watershed *	% of patch	USGS 2003
Mean Canopy Cover in sample watershed	% of patch	USGS 2009
Mean Canopy Cover riparian area of sample watershed	% of patch	USGS 2009
% Forest Area in sample watershed*	% of patch	USGS 2009
Land use Area by Category in sample watershed	ha and % of patch	USGS 2009
Land use Area by Category in sample watershed	ha and % of patch	USGS 2009
Geology Type and category in sample watershed		DMME 2008

50 patches from the 9 groups. In each selected patch we placed two pairs (air and water) of thermographs, one at the patch pour point and one at the centroid. Because of dry stream channels (primarily centroids) and lost or stolen thermographs, we had a complete data set on 77 watersheds (pour points = 43; centroids = 34).

We focused on metrics associated with daily maximum water temperature during the critical period for this study because

(1) increases in air temperature (and presumed increases in water temperature) have the highest probability occurrence in various climate change scenarios (Intergovernmental Panel on Climate Change 2007),

(2) daily maximum water temperature metrics for presence and absence of reproducing populations of brook trout are known (Stoneman and Jones 1996; Picard et al. 2003; Wehrly et al. 2003; Huff et al. 2005; Wehrly et al. 2007) and

(3) we believe lethal water temperature effects from climate change will likely occur first and have an immediate and dramatic effect on existing brook trout populations.

Sampling Protocol

Paired (air and water) thermographs (HOBO Watertemp Pro v2; accuracy 0.2˚C; drift <0.1 annually; Onset Computer Corporation 2008) were placed at the pour point and at the centroid of each sampled patch. All thermographs were set to record every 30 min (Dunham et al. 2005; Huff et al. 2005) from July 1st through September 30th (Stoneman and Jones 1996), thus encompassing the only period when water temperatures are likely to exceed the lethal limit (> 21ºC) for brook trout in Virginia.

Thermographs were calibrated pre and post-deployment following methods summarized in Dunham et al. (2005). Because of the possibility that stream channels may run dry during summer low flow periods, thermographs used to record water temperatures were placed near maximum residual pool depths (Lisle 1987) when possible. A shield was used to reduce direct UV contact with air temperature thermographs (Dunham et al. 2005; Wise et al. 2010).

Metrics

We define sensitivity as the change in the daily maximum water temperature (DMAXW) from a 1°C increase in the daily maximum air temperature (DMAXA). Because this sensitivity varies throughout the DMAXA range we report the median change for each watershed as a single sensitivity metric.

In addition, we developed a standardized vulnerability score for each brook trout habitat patch from the DMAXW values. Duration (number of consecutive days above 21°C), frequency (proportion of days above 21°C) and magnitude (average DMAXW of all DMAXW days over 21°C) for the sample period were standardized ((x-mean)/SD) and the average of the three standardized metrics was used for the final vulnerability score. We view this as the "effective dose" associated with increased water temperature. Combining the sensitivity scores with the vulnerability scores resulted in four classification categories: (high sensitivity-high vulnerability (HS-HV); high sensitivity-low vulnerability (HS-LV); low sensitivity-high vulnerability (LS-HV) and low sensitivity-low vulnerability (LS-LV). The cutoffs used in this classification were: > 0.38 °C high sensitivity, <= 0.38 °C low sensitivity; vulnerability > -0.75 high vulnerability, < -0.75 low vulnerability

Results

The response of DMAXW to a 1°C increase in DMAXA averaged 0.38 °C among all sites and air temperature ranges (Figure 2). However, there was considerable variation. For example a one degree DMAXA increase from 16 to 17 °C averaged a 0.52 °C increase in DMAXW but ranged from 0.13 to 0.98 °C dependent on the sample site. A 1°C increase in DMAXA from 25 to 26 °C averaged (0.35) with a range of 0.10 to 0.82 °C (Figure 2).

Our vulnerability metrics also varied among patches; duration averaged 11.75 d (SD = 17.1; range 0 to 56 d); frequency averaged 23% (SD = 29.0%; range 0.0 to 91.0%) and magnitude above 21°C averaged 0.76 (SD = 0.99; range 0.00 to 3.80).

The predicted maximum air temperature from the PRISM (2007) data at the sample location was correlated with the vulnerability metrics ($r = 0.55$ duration; $r = 0.52$ proportion; $r = 0.49$ magnitude) as was the sample location elevation ($r = -0.57$ duration; $r = -0.56$ proportion; $r = -0.53$ magnitude). Neither the predicted maximum air temperature ($r = 0.05$) nor elevation ($r = -0.06$) at the sample locations were correlated with the sensitivity metric. The solar insolation in the riparian area (JUL 1 to SEP 30) corrected for canopy cover was not as highly correlated with the vulnerability metrics ($r = 0.13$ duration; $r = 0.23$ proportion; $r = 0.20$ magnitude) as it was with the sensitivity metric ($r = 0.28$).

Our sensitivity-vulnerability classification system categorized 51.9% of the brook trout habitat patches as both sensitive and vulnerable (HS-HV) to climate change followed by 29.9% (LS-LV); 10.4% (HS-LV) and 7.8% LS-HV (Figure 3). There were significant differences among the group means for sample point elevation (ANOVA, $F = 4.44$; $df = 73$; $P < 0.006$); sample point predicted maximum air temperature (ANOVA, $F = 4.76$; $df = 73$; $P < 0.004$); the watershed area above the sample points (ANOVA, $F = 5.10$; $df = 73$; $P < 0.003$) and the average solar gain (30 m pixel)(JUL 1 to SEP 30) in the riparian area corrected for canopy cover (ANOVA, $F = 4.89$; $df = 73$; $P < 0.004$). In general HS-HV patches were found at lower elevations, with larger watershed areas and higher solar insolation values than LS-LV patches.

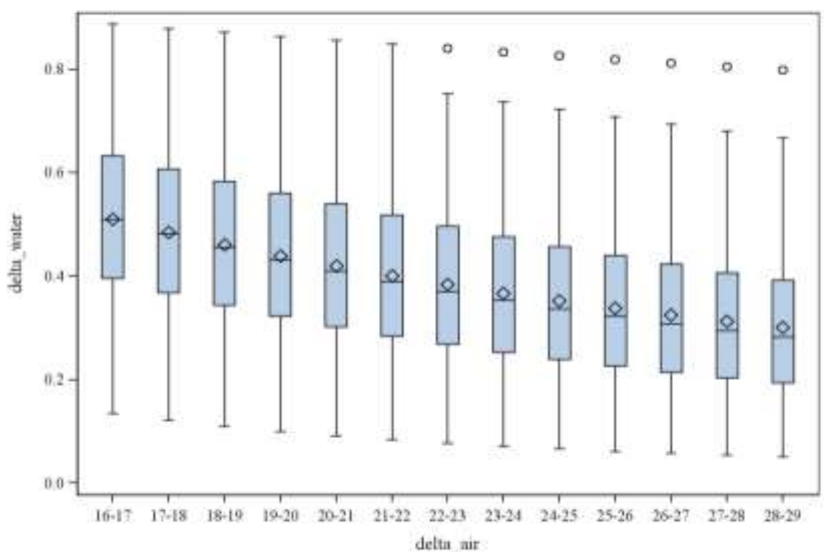


Figure 2. Box plots of the response of daily maximum water temperatures to a 1°C increase in daily maximum air temperature (by one degree bins from 16 to 28 °C) from 77 patches of brook trout habitat during July 1 through September 30, 2009.

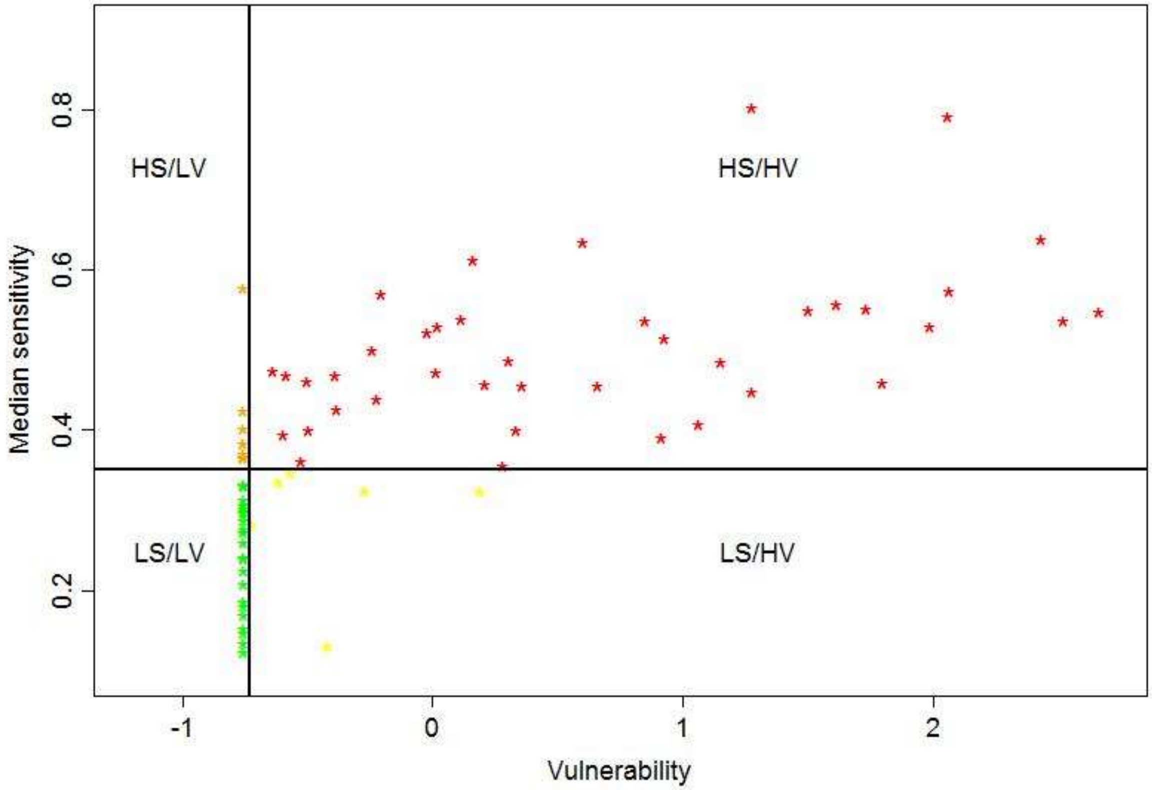


Figure 3. Sensitivity and Vulnerability Classification Chart. (High Sensitivity-High Vulnerability = HS-HV; High Sensitivity-Low Vulnerability = HS-LV; Low Sensitivity-High Vulnerability = LS-HV; Low Sensitivity-Low Vulnerability = LS-LV). See methods for quadrant thresholds.

Discussion

Our direct measurement approach will produce markedly different predictions of future brook trout distributions than models that used secondary data to predict the relationship between air and water temperatures. In most cases, our direct measurement approach identified more brook trout watersheds that are not sensitive and currently not vulnerable to predicted air temperature increases. While typical secondary data sources (maximum air temperature and elevation) used in regional models to predict water temperature were correlated with our sensitivity and vulnerability metrics, our direct measurement approach reduces the chances of error. Air temperature increases from the various climate change scenarios can be applied to the watershed specific air-water temperature relationship curves instead of secondary data model averages to better predict vulnerability to extirpation.

We found considerable variability in our sensitivity metrics among brook trout habitats. Land use metrics such as the interaction of aspect, solar insolation, and canopy cover used in our study may explain the residual errors in these larger models and provide more information for mangers (Fu and Rich 1999; Fu and Rich 2002).

We recommend that when managers make long term planning decisions, such as choosing among populations of brook trout for preserving genetic information or for making investments in habitat restoration, that they develop site specific air-water temperature relationships instead of relying on existing secondary data models.

Combined with our sensitivity-vulnerability classification system this direct measurement approach gives managers a tool to assess potential persistence of individual brook trout habitats under various climate change scenarios. We recommend their use when the potential for costs of an error (either Type 1 or Type 2) from the secondary data models are high.

Although climate change effects other than air temperature (i.e. rainfall, floods, droughts, changes in land cover, spawning times, invasive species, etc.) are important, the low predictability of these metrics (both in magnitude and direction) at this time make it difficult for managers to incorporate this information into the decision making process. Predictions of increasing air temperatures have the highest reliability (Intergovernmental Panel on Climate Change 2007), and we believe that these increases pose the highest risk of change to the current distribution of brook trout. This study is currently being expanded to encompass the entire range of brook trout habitat in the southeastern United States.

Acknowledgements

We would like to thank the Virginia Department of Game and Inland Fisheries and the George Washington and Jefferson National Forest for purchasing the thermographs for this study. A special thanks to L. Mohn, P. Bugas, S. Reeser, J. Hallacher, S. Coffman and D. Kirk for support of this study. We thank Z. Robinson; C. Kyger; A. Fitzgerald, and L. Wise for deploying and collecting the thermographs. We thank P. Flebbe and N. Schmal for earlier reviews of this work.

References

Clark, M. E., K. A. Rose, D. A. Levine, and W. W. Hargrove. 2001. Predicting climate change effects on Appalachian brook trout: Combining GIS and individual-based modeling. Ecological Applications 11:161-178.

Cluis, D. A. 1972. Relationship between stream water temperature and ambient air temperature: A simple autoregressive model for mean daily stream water temperature fluctuations. Nordic Hydrology 3:65-71.

DMME (Virginia Department of Mines, Minerals, and Energy). 2008. General Geologic Map of Virginia. Available: http://www.dmme.virginia.gov/DMR3/dmrpdfs/VA%20geol%20map.pdf (July 2010)

Dunham, J., C. Gwynne, B. Reiman, and D. Martin. 2005. Measuring stream temperature with digital data loggers: A user's guide. USDA Forest Service, Report RMRS-GTR-150WWW, Fort Collins, CO.

EBTJV (Eastern Brook Trout Joint Venture). 2006. http://www.easternbrooktrout.org (July 2010)

ESRI. 2009. Solar radiation, extension of spatial analyst tools. Redlands, California

Fink, D. B. 2008. Artificial shading and stream temperature modeling for watershed restoration and brook trout (*Salvelinus fontinalis*) management. Master's thesis. James Madison University, Harrisonburg, Virginia.

Flebbe, P. A. 1994. A regional veiw of the margin: Salmonid abundance and distribution in the southern Appalachian mountians of North carolina and Virginia. Transactions of the American Fisheries Society 123:657-667.

Flebbe, P. A., L. D. Roghair, and J. L. Bruggink. 2006. Spatial modeling to project southern Appalachian trout distribution in a warmer climate. Transactions of the American Fisheries Society 165:1371-1382.

Fu, P. and P.M. Rich. 1999. Design and implementation of the Solar Analyst: An Arcview extension for modeling solar radiation at landscape scales. Proceedings of the 19th Annual ESRI User Conference, San Diego, California. http://esri.com/library/userconf/proc99/proceed/papers/pap867/p867.htr

Fu, P. and P.M. Rich. 2002. A geometric solar model with applications in agriculture and forestry. Computers and Electronics in Agriculture. Volume 37(1-3): 25-35.

Hudy, M., T. M. Thieling, N. Gillespie, and E. P. Smith. 2008. Distribution, status, and land use characteristics of subwatersheds within the native range of brook trout in the eastern United States. North American Journal of Fisheries Management 28:1069-1085.

Huff, D.D., S.L. Hubler, and A.N. Borisenko. 2005. Using field data to estimate the realized thermal niche of aquatic vertebrates. North American Journal of Fisheries Management 25: 346-360.

Intergovernmental Panel on Climate Change. 2007. Climate Change 2007: The Physical Science Basis – Summary of Policy makers. http://www.ipcc.ch/SPM2feb07.pdf

Isaak, D. J., and W. A. Hubert. 2001. A hypothesis about factors that affect maximum summer stream temperatures across montane landscapes. Journal of the American Water Resources Association 37:351-366.

Johnson, S. L. 2003. Stream temperature: Scaling of observations and issues for modelling. Hydrological Processes 17:497-499.

Lisle, T. E. 1987. Using "residual depths" to monitor pool depths independently of discharge. US Forest Service Research Note PSW-394

Keleher, C. J., and F. J. Rahel. 1996. Thermal limits to salmonid distributions in the rocky mountain region and potential habitat loss due to global warming: A geographic information systems (GIS) approach. Transactions of the American Fisheries Society 125:1-13.

Meisner, J. D. 1990. Effect of climate warming on the southern margins of the native range of brook trout, salvelinus fontinalis. Canadian Journal of Fisheries and Aquatic Science 47:1065-1070.

Mohn, L. and P. E. Bugas, Jr. 1980. Virginia trout stream and environmental inventory. Federal Aid in Fish Restoration, Project F-32, final report. Virginia Department of Game and Inland Fisheries. Richmond, Virginia.

Mohseni, O., and H. G. Stefan. 1999. Stream temperature/air temperature relationship: A physical interpretation. Journal of Hydrology 218:128-141.

Moore, R. D., D. L. Spittlehouse, and A. Story. 2005. Riparian microclimate and stream temperature response to forest harvesting: A review. Journal of the American Water Resources Association :813-834.

Onset Computer Corporation. 2009. HOBO U22 Water Temp Pro v2 users manual. Document # 10366-C. http://www.onsetcomp.com/files/manual_pdfs/10366-C-MAN-U22-001.pdf

Picard, C. R., M. A. Bozek, and W. T. Momot. 2003. Effectiveness of using summer thermal indices to classify and protect brook trout streams in northern Ontario. North American Journal of Fisheries Management 23:206-215.

Pilgrim, J. M., X. Fang, and H. G. Stefan. 1998. Stream temperature correlations with air temperatures in Minnesota: Implications for climate warming. Journal of the American Water Resources Association 34(5):1109-1121.

PRISM. 2007. Title. Oregon State University, Corvallis. Available: http://www.prism.oregonstate.edu/. (November 2009).

Rahel, F. J., C. J. Keleher, and J. L. Anderson. 1996. Potential habitat loss and population fragmentation for coldwater fish in the north platte river drainage of the rocky mountinas: Response to climate warming. Limnology and Oceanography 41(5):1116-1123.

Rieman, B. E., D. Isaak, S. Adams, D. Horan, D. Nagel, C. Luce, and D. Meyers. 2007. Anticipated climate warming effects on bull trout habitats and populations across the interior columbia river basin. Transactions of the American Fisheries Society 136:1552-1565.

SAS. 2000. SAS/STAT User’s Guide, Version 8(1). SAS Institute, Cary

Stoneman, C. L., and M. L. Jones. 1996. A simple method to classify stream thermal stability with single observations of daily maximum water and air temperatures. North American Journal of Fisheries Management 16:728-737.

USGS (United States Geological Survey) 2008. The national map seamless server. http://seamless.usgs.gov/website/seamless/viewer.htm. Accessed 24 September, 2008

USGS (United States Geological Survey) 2009. National land cover dataset 2001. USGS, Washington, D.C. Available: www.mrlc.gov. (February 2009).

USGS (United States Geological Survey) 2003. Baseflow index grid for the conterminous United States. USGS, Washington, D.C. Available: http://water.usgs.gov/lookup/getspatial?bfi48grd (February 2009).

Wehrly, K.E., L. Wang, and M. Mitro. 2007. Field-based estimates of thermal tolerance limits for trout: Incorporating exposure time and temperature fluctuation. Transactions of the American Fisheries Society 136:365-374.

Wehrly, K. E., M. J. Wiley, and P. W. Seelbach. 2003. Classifying regional variation in thermal regime based on stream fish community patterns. Transactions of the American Fisheries Society 132:18-38.

Williams, J. E., A. L. Haak, H. M. Neville, and W. T. Colyer. 2009. Potential consequences of climate change to persistence of cutthroat trout populations. North American Journal of Fisheries Management 29:533-548.

Wise, L. M., B. A. Trumbo and M. Hudy. 2010. Determining Drift in Electronic Temperature Sensors Used in Climate Change Modeling. James Madison University undergraduate research (unpublished).

PREDICTED EFFECTS OF CLIMATE CHANGE ON THE DISTRIBUTION OF WILD BROOK TROUT AND BROWN TROUT IN WISCONSIN STREAMS

Matthew G. Mitro, Fisheries Research Scientist
Wisconsin Department of Natural Resources, 2801 Progress Road, Madison, WI 53716, Phone: (608) 221-6366 Fax: (608) 221-6353 Email: matthew.mitro@wisconsin.gov

John D. Lyons, Fisheries Research Scientist
Wisconsin Department of Natural Resources, 2801 Progress Road, Madison, WI 53716, Phone: (608) 221-6328 Fax: (608) 221-6353 Email: john.lyons@wisconsin.gov

Jana S. Stewart, Geographer
U.S. Geological Survey, 8505 Research Way, Middleton, WI 53562, Phone: (608) 821-3855 Fax: (608)821-3817 Email: jsstewar@usgs.gov

ABSTRACT — Climate change is expected to result in increases in summer air and water temperatures in Wisconsin, but impacts on the distribution of wild trout in streams therein are unknown. Understanding potential climate warming impacts on wild trout will help fisheries managers in developing adaptation strategies to best protect and enhance coldwater resources confronted by climate change. We developed models to estimate the potential effects of climate warming on the distribution of brook trout *Salvelinus fontinalis* and brown trout *Salmo trutta* in Wisconsin streams. We used classification tree analyses, 69 environmental variables in a geographic information system, and fish surveys from 1,628 stream locations to predict the distribution of brook trout and brown trout in all stream reaches in Wisconsin (1:100,000 scale; 35,748 stream reaches totaling 86,958 km in length). We applied the models to current climate conditions and three warming scenarios based on regional-scale climate model projections for Wisconsin: summer temperature increases of 1°C (air) and 0.8°C (water); 3°C (air) and 2.4°C (water); and 5°C (air) and 4°C (water). The models indicated that under current climate conditions 28,802 stream km (33%) are suitable for brook trout and 37,241 stream km (43%) are suitable for brown trout. Significant losses in stream habitat (km) suitable for trout are predicted to occur under all three warming scenarios. Predicted changes in brook trout distribution are -44%, -94%, and -100% under the three ascending warming scenarios. Corresponding predicted changes in brown trout distribution are -8%, -33%, and -88%. Regionally, losses are predicted to initially be most dramatic in small streams in northern Wisconsin and less dramatic in the Driftless Area of southwestern Wisconsin. Possible adaptation strategies to lessen the impact of climate warming effects on trout include land, water, and riparian management strategies, stream restoration, and a triage approach to stream management.

INTRODUCTION

The native brook trout *Salvelinus fontinalis* and the introduced brown trout *Salmo trutta* are the most common wild trout found in Wisconsin streams. Water temperature is a critical factor in determining where trout can live, and hence, how they are distributed across streams in Wisconsin. Summer water temperature can be used to classify streams across a spectrum from cold water to warm water. Wisconsin streams that maintain relatively cold summer maximum water temperatures and provide habitat for trout include streams that are classified as coldwater or cold transition streams (Lyons et al. 2009). These streams have water temperatures <18.7°C (June-August mean), <19.5°C (July mean), and <22.6°C (maximum daily mean) (Lyons et al. 2009).

Streams in Wisconsin currently comprise 7.9% coldwater, 45.9% cold transition, 28.6% warm transition, and 17.6% warmwater stream kilometers (86,958 total stream km; 1:100,000 scale U.S. Hydrography Data; Lyons et al. 2009). Both coldwater and cold transition streams can support high abundances of coldwater fishes such as trout, with brown trout abundance often highest in cold transition streams. Coldwater fishes are less abundant in warm-transition streams and essentially absent in warmwater streams during summer. Trout are sensitive to changes in environmental conditions, particularly water temperature. As such, they may be particularly susceptible to the effects of climate change on streams. Changes in maximum water temperatures of only a few degrees can have significant effects on the occurrence and distribution of trout in streams (Magnuson et al. 1997; Lyons et al. 2009).

Air temperature is a significant determinant of summer maximum water temperature in streams (Pilgrim et al. 1998; Stewart et al. 2006). Climate change is expected to result in increases in summer air temperatures in Wisconsin, with projected increases in annual air temperatures of 1to 8 °C by the year 2050 in the Great Lakes region of North America (Magnuson et al. 1997). If these projected increases in air temperature are realized, we can expect summer maximum water temperatures in streams to increase as well (Pilgrim et al. 1998; Mosheni et al. 1999). However, the relation between air temperature and water temperature is complex, as is evident by the occurrence of highly heterogeneous stream temperatures as compared to air temperature across the landscape. Stream reaches may exhibit different temperature regimes in close proximity to one another and under the same climate conditions because of local variation in geology and groundwater contributions to streams.

The impacts climate change will have on the distribution of wild trout in Wisconsin streams are unknown. Understanding the potential impacts of climate warming on wild trout will help fisheries managers to develop adaptation strategies to best protect and enhance coldwater fisheries confronted by climate change. We developed models to estimate the potential effects of climate warming on the distribution of wild brook trout and brown trout in Wisconsin streams. The models include both non-thermal stream, riparian, and watershed-scale habitat factors and dynamic climate factors to characterize small-scale variation in stream conditions across Wisconsin. We used these models to estimate the effects of three climate warming scenarios on the distribution of wild trout in Wisconsin streams, and we discuss practicable adaptation strategies fisheries managers can use to ameliorate climate change impacts on wild trout fisheries in streams.

Methods

We developed models using fish and environmental data collected from 393 sites located on 282 Wisconsin streams and rivers to predict the changes in coldwater habitat and trout distribution in Wisconsin streams that might occur under different climate warming scenarios. We tested the models using independent fish data not used in model development. The sampling sites were located throughout the state and included a wide range of environmental conditions. We sampled each site once using electrofishing during summer low flows between 1995 and 1999 and collected all species of fish encountered. Sites varied in length depending on the width of the stream. See Lyons et al. (in press) for further details on fish sampling methods.

We ran models for each stream reach in the state for brook trout and brown trout under current climate conditions and three climate warming scenarios specific to Wisconsin. We used streams mapped at the 1:100,000 scale (U.S. Hydrography Data) and divided into 35,748 discrete segments (86,958 total stream km; 2.43 km mean segment length). We determined and assigned 27 environmental characteristics to each segment using GIS. The 27 variables were based on previous studies that identified important environmental factors describing fish distributions in Wisconsin. These variables described, at scales ranging from the riparian zone to the watershed, the topography, land cover, watershed size and position, surficial and bedrock geology, potential groundwater inputs, and stream network characteristics for a total of 58 variables.

We used some of these environmental variables along with climate variables such as air temperature and precipitation to estimate 11 additional variables that describe stream flow and temperature for each stream segment. Stream flow variables were estimated from an unpublished stream flow model (L. Hinz, Illinois Department of Natural Resources, Springfield, U.S.A.), and stream temperature variables were estimated from an artificial neural network model (Stewart et al. 2006).

We estimated statewide habitat suitability of streams for brook trout and brown trout using watershed-scale regression tree models. Model variables were selected from the 69 GIS-based environmental variables, some of which were based on climate variables. Model outputs included predictions of the presence or absence of each trout species in every stream segment in Wisconsin. We then constrained model predictions in four ways to provide more ecologically realistic predictions. The four constraint variables included location in the state, water temperature, stream size, and land use variables. These variables are important determinants of fish distribution in Wisconsin, but may not have been present in final models. For example, if trout were predicted to be present in a stream segment in which the modeled water temperature was clearly unsuitable for trout (e.g., average maximum daily mean water temperature > 24.6°C), then the segment was changed from present to absent. We applied the final trout species models to each stream segment in the state using the appropriate environmental variables to determine whether or not each trout species was predicted to be present or absent, and we mapped the results. Additional details on model variables, development, and constraints are in Lyons et al. (in press).

The climate warming scenarios came from climate model projections for Wisconsin, obtained from the University of Wisconsin-Madison Climate Working Group of the Wisconsin Initiative on Climate Change Impacts (unpublished data available on the web at http://www.wicci.wisc.edu/climate-change.php and http://ccr.aos.wisc.edu/cwg; accessed 30 July 2010). The Climate Working Group 'downscaled' continental climate predictions from global circulation models (GCM) from 150-km-square grids to 10-km-square grids, resulting in more specific predictions for Wisconsin. We considered three climate change scenarios of the 45 generated by the Climate Working Group using 15 GCM with 3 emission scenarios over 50 years. The three scenarios addressed increases in the average summer June-July air temperature, with a corresponding increase in stream water temperature at a rate of 0.8 times the increase in air temperature, a rate derived for streams in the adjacent state of Minnesota (Pilgrim et al. 1998). The climate warming scenarios include the following temperature increases: (1) a limited increase of 1°C air and 0.8°C water (Global Circulation Model CSIRO-M3K-0 under optimistic projections of future greenhouse gas emissions); (2) a moderate increase of 3°C air and 2.4°C water (Global Circulation Model GISS-AOM with "business as usual" projections of future greenhouse gas emissions); and (3) a major increase of 5°C air and 4°C water (Global Circulation Model MIROC3-HIRES with "business as usual" projections of future greenhouse gas emissions; Lyons et al. in review). The limited increase represented the smallest increase of the 45 GCM-emission predictions, and the moderate and major increases represented the mean and second highest increases. We re-ran each of the trout species models using the warmer air and water temperatures projected in each of the three climate warming scenarios. Additional details on the climate warming scenarios are in Lyons et al. (in press).

Note that we did not consider changes in precipitation in the climate change scenarios. The stream temperature models currently cannot account for effects of precipitation on groundwater recharge and input to streams and hence to stream temperature. We are currently working with scientists at USGS (Middleton, Wisconsin) to update the stream temperature models using a soil-water-balance modeling approach to better link precipitation to groundwater and stream temperature. We also held land use constant at current conditions in each modeling run.

Results

The regression tree model for brook trout had ten branches and included an air temperature variable. The first split or branching point was the most important variable explaining the distribution pattern, which for brook trout was the 50% exceedence flow (median flow) for the entire year ($m^3 \cdot s^{-1}$). The second split or next most important variable in the brook trout model was mean annual air temperature (°C) in the upstream catchment. The regression tree for brown trout had three branches and did not include any air or water temperature variables. Therefore, predicted declines in brown trout resulted from the water temperature constraint on model predictions. The first split for brown trout was the 10% exceedence flow (high flow) for the month of April ($m^3 \cdot s^{-1}$) and the second split was the percent land area in the upstream catchment with sandstone bedrock. Model testing results are presented in Table 1.

Table 1. Overall percentage accuracy, and percentage correct prediction of absence (specificity) and of presence (sensitivity) for brook trout and brown trout model development and validation datasets.

	Development			Validation		
	Overall	Absence	Presence	Overall	Absence	Presence
Fish species	%	%	%	%	%	%
Brook trout	87.8	86.1	98.2	67.0	64.5	71.6
Brown trout	79.9	82.1	69.6	64.3	53.3	89.3

Table 2. Predicted stream length (km) and percentage of the total stream length in Wisconsin (86,958 km) suitable for brook trout and brown trout under current air and water temperatures, and predictions of the lengths of suitable stream and percentage change from current climate conditions under three climate warming scenarios.

Climate	Brook trout		Brown trout	
Air °C (Water °C)	Stream length (km)	% of total	Stream length (km)	% of total
Current	28,802	33	37,241	43
		% change		% of total
+1 (+0.8)	16,245	-44	34,296	-8
+3 (+2.4)	1,618	-94	24,908	-33
+5 (+3.0)	0	-100	4,378	-88

The models indicated that under current climate conditions 28,802 stream kilometers (33%) were suitable for brook trout and 37,241 stream kilometers (43%) were suitable for brown trout (Table 2, Figures 1and 2). Substantial declines in both trout species were predicted with increasing air and water temperatures. Significant losses in stream habitat (km) suitable for trout are predicted to occur under all three warming scenarios, with brook trout extirpated from Wisconsin streams under the major climate warming scenario. Predicted changes in brook trout distribution were -44%, -94%, and -100% under the three ascending warming scenarios. Corresponding predicted changes in brown trout distribution were -8%, -33%, and -88%. Regionally, losses were predicted to initially be most dramatic in small streams in northern Wisconsin and less dramatic in the Driftless Area of southwestern Wisconsin.

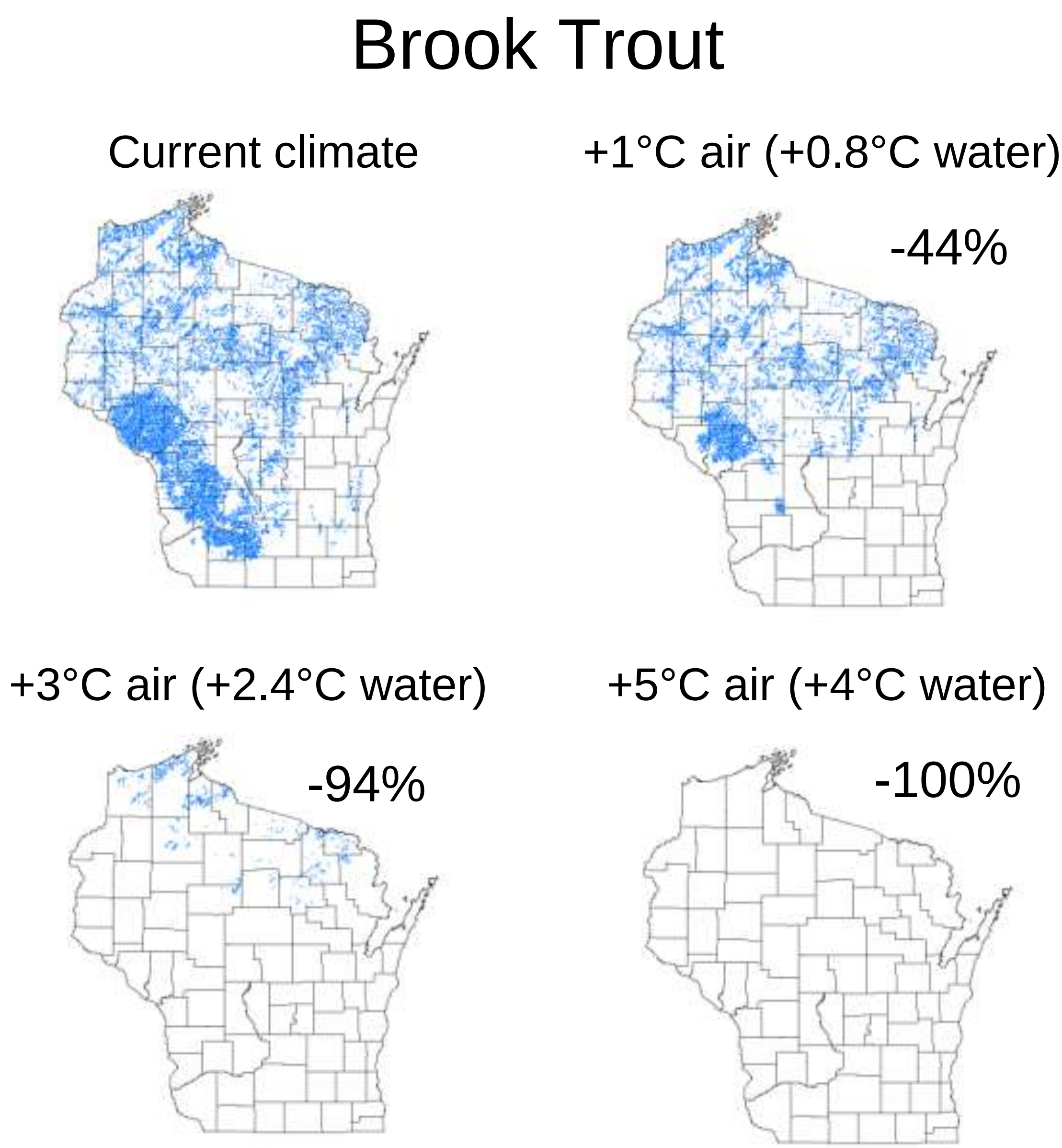


Figure 1. Predicted distribution of brook trout in Wisconsin streams under current climate conditions and three climate-warming scenarios (change in average June-August daily mean air and water temperature in streams). Percentages show loss of stream length from the current distribution suitable for brook trout.

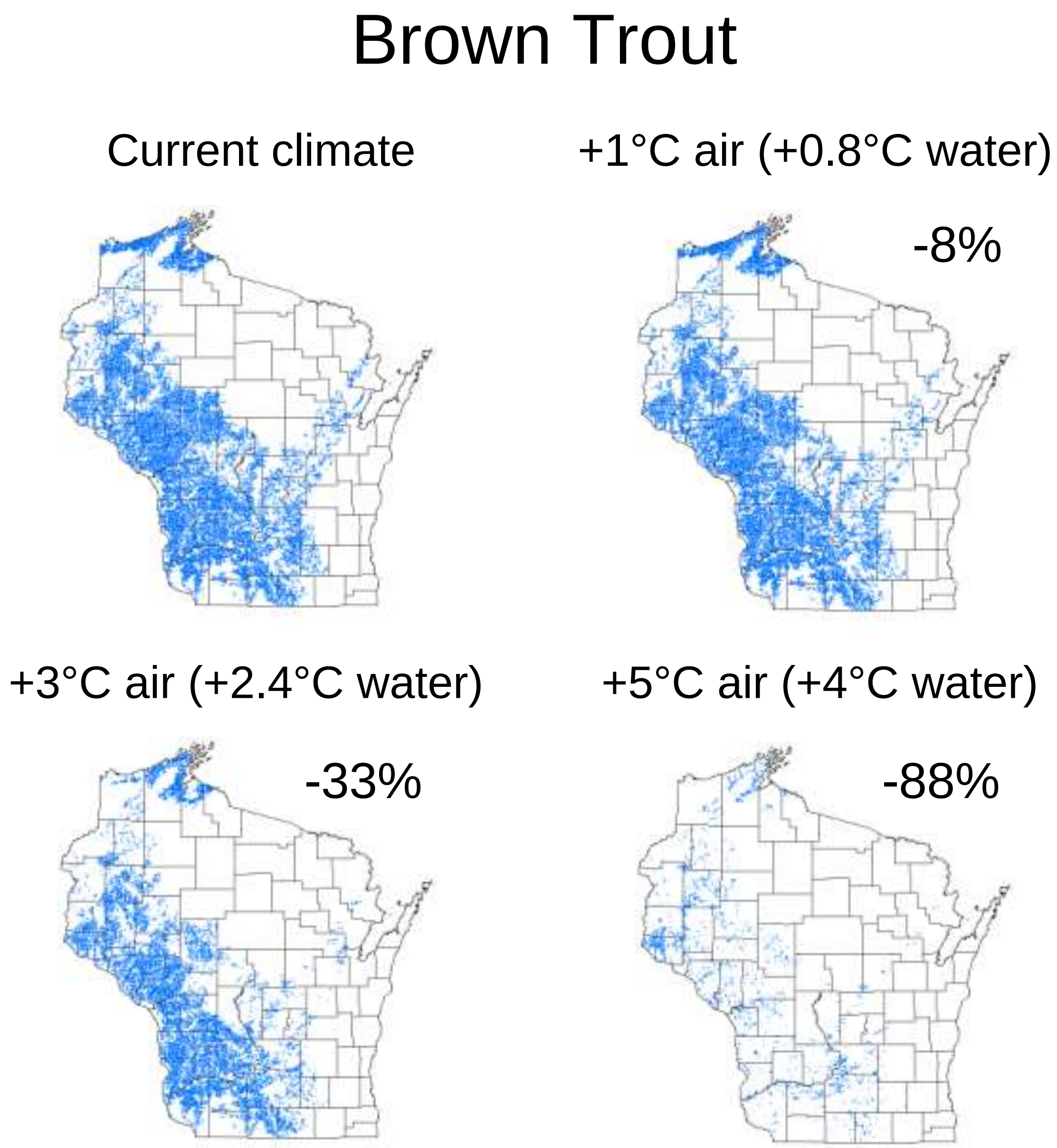


Figure 2. Predicted distribution of brown trout in Wisconsin streams under current climate conditions and three climate-warming scenarios (change in average June-August daily mean air and water temperature in streams). Percentages show loss of stream length from the current distribution suitable for brown trout.

Discussion

We found that statistical models using watershed-scale environmental variables in a GIS framework can accurately predict trout distribution in Wisconsin streams. Environmental variables that were most useful in the Wisconsin trout models were related to stream size (catchment area and stream flow), climate (summer air temperatures), and summer water temperature. The overall accuracy, specificity, and sensitivity of our models were similar or better than values for comparable fish models developed for other temperate areas (see references in Lyons et al., in press). The trout models developed in this study provided valuable insights into how the distribution of wild trout may change under different climate warming scenarios, in which climate warming could lead to major declines.

Introduced wild brown trout are currently more widespread than native brook trout in Wisconsin streams. For all three climate change scenarios both species declined, brown trout the least and brook trout the most. Losses of habitat were expected to occur evenly across the state and were not noticeably concentrated in any particular geographic region. Under limited or moderate increases in air and water temperatures, some stream habitat that becomes unsuitable for brook trout may remain or become suitable for brown trout, thus lessening climate-induced losses of brown trout relative to brook trout.

Model results clearly indicate that climate change has the potential to cause major declines in stream trout distribution in Wisconsin, including possible extirpation. However, three caveats should be considered when interpreting model predictions. First, the models assume that water temperatures will increase similarly in all streams across the state in response to rising air temperatures. This assumption is an oversimplification but is required to allow the climate warming scenarios to run in the models, given the large number of stream segments (35,748) and the complexity of the stream temperature neural network model. In reality, for any given climate change scenario, the response of water temperature to air temperature will vary depending on geographic location and stream segment characteristics such as ground water input, channel morphology, shading, and land use. Some stream reaches predicted to become unsuitable for wild trout may in fact remain suitable, with the models possibly overstating total losses. We are currently developing improved versions of the models that allow for geographic and reach-specific changes in water temperature.

A second caveat is the absence of a link in the models between changes in precipitation, groundwater recharge, and groundwater input to streams, and hence, water temperature. We therefore could not include changes in precipitation amount and pattern in the three climate change scenarios. Future climate scenarios for Wisconsin differ in how precipitation is expected to change. Some project increases and others project decreases, but few project that precipitation patterns will remain unchanged. Increases in precipitation could enhance groundwater inputs and moderate potential water temperature increases caused by warmer air temperature. Declines in wild trout may consequently be moderated. Decreases in precipitation could have the opposite effect. Many climate change scenarios indicate flooding and drought are likely to become more common in the future, and such events can potentially harm wild trout. Future improvements to the models will include capabilities to incorporate variation in precipitation and groundwater inputs across the state for use in predicting water temperature in streams.

A final caveat is that we ran the models under the unrealistic assumption that land use will remain unchanged over the next 50 years. Land use is constantly changing, and several land-use trends of recent decades are expected to continue into the future. These trends could have several important consequences for coldwater streams that were not incorporated into the model predictions. First, urban areas will probably continue to expand, with negative effects of increasing stream temperature and decreasing trout populations (Wang et al. 2003; Stewart et al. 2006). Second, in some areas of the state such as southwestern Wisconsin, agricultural lands may continue to be converted to fallow fields, woodlands, or other less intensive uses, generally to the benefit of coldwater species (Wang et al. 2002; Marshall et al. 2008). In other areas, such as east-central and north-central Wisconsin, agricultural land use may intensify, generally to the detriment of coldwater species. About 39% of current groundwater withdrawal is used for irrigation, and such use of groundwater may increase with agricultural land use. Finally, given that many coldwater streams support popular and valuable sport fisheries for wild trout, riparian and watershed land management activities designed specifically to benefit trout are widespread and likely to continue and perhaps expand in an effort to lessen impacts of warmer air temperatures on coldwater streams. In future modeling runs, we will explore the relative effects of these trends in land-use in combination with climate change on coldwater streams and wild trout.

Management Implications

Climate-induced changes in stream temperature and flow will not be uniform. Interactions between air temperature and precipitation and stream temperature and flow are mediated by stream channel, riparian, and watershed characteristics. It follows that the ability of streams to buffer change in water temperature and flow against change in climate will vary. Herein lays opportunity for managing climate impacts on inland trout and other coldwater resources. We suggest two types of adaptation strategies that can be used to lessen the impact of climate warming effects on trout. The first involves environmental management activities to offset the impacts of rising air temperatures and changes in precipitation. These activities include land, riparian, and water management and stream restoration. The second involves a triage approach to identifying potential impacts of climate change to coldwater resources and allocating management resources to those coldwater habitats most likely to realize success. Some streams, for example, may face inevitable losses of coldwater fishes, some may be resilient to climate impacts, and some may allow for persistence of coldwater fishes contingent on management approaches used to counteract climate impacts. Appropriate management actions may include environmental adaptation strategies as well as changes to angling regulations and fish stocking strategies. We expect that a proactive application of these adaptation strategies will help protect Wisconsin's coldwater fishes and fisheries from the impacts of our changing climate.

Acknowledgements

We thank T. Brenden, A. Cooper, L. Hinz, P. Kanehl, A. Martin, C. Smith, and S. Westenbroek for helping collect, compile, and process the data used to generate the models and stream segment characteristics. U.S. Environmental Protection Agency 'Science to Achieve Results' Grant (R830596) through the National Center for Environmental Research; the U.S. Geological Survey, National GAP Analysis Program, Great Lakes Aquatic Gap Project; Federal Aid in Sport Fish Restoration Project F-80-R, Project F-95-P, studies SSMP and SSCN; and the Wisconsin Department of Natural Resources provided support for this study.

References

Lyons, J., T. Zorn, J. Stewart, P. Seelbach, K. Wehrly, and L. Wang. 2009. Defining and characterizing coolwater streams and their fish assemblages in Michigan and Wisconsin, USA. North American Journal of Fisheries Management 29:1130-1151.

Lyons, J. D., J. S. Stewart, and M. G. Mitro. In press. Predicted effects of climate warming on the distribution of 50 stream fishes in Wisconsin, U.S.A. Journal of Fish Biology.

Magnuson, J. J., K. E. Webster, R. A. Assel, C. J. Bowser, P. J. Dillon, J. G. Eaton, H. E. Evans, E. J. Fee, R. I. Hall, L. R. Mortsch, D. W. Schindler, and F. H. Quinn. 1997. Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian Shield Region. Hydrological Processes 11:825-871.

Marshall, D. W., A. H. Fayram, J. C. Panuska, J. Baumann, and J. Hennessy. 2008. Positive effects of agricultural land use changes on coldwater fish communities in southwest Wisconsin streams. North American Journal of Fisheries Management 28:944-953.

Mohseni, O., D. L. Erickson and H. G. Stefan. 1999. Sensitivity of stream temperatures in the United States to air temperatures projected under a global warming scenario. Water Resources Research 35:3723-3733.

Pilgrim, J. M., X. Fang, and H. G. Stefan. 1998. Stream temperature correlations with air temperatures in Minnesota: implications for climate warming. Journal of the American Water Resources Association 34:1109-1121.

Stewart, J., M. Mitro, E. A. Roehl, Jr., and J. Risley. 2006. Numerically optimized empirical modeling of highly dynamic, spatially expansive, and behaviorally heterogeneous hydrologic systems – part 2. In Proceedings of the 7th International Conference on Hydroinformatics, Nice, France. pp. 1-8.

Wang, L., J. Lyons, and P. Kanehl. 2002. Effects of watershed best management practices on habitat and fish in Wisconsin streams. Journal of the American Water Resources Association 38:663-680.

Wang, L., J. Lyons, and P. Kanehl. 2003. Impacts of urban land cover on trout streams in Wisconsin and Minnesota. Transactions of the American Fisheries Society 132:825-839.

LONG-TERM STUDY OF BROOK TROUT ABUNDANCE, GROWTH, AND MOVEMENT IN THE STAUNTON RIVER, SHENANDOAH NATIONAL PARK, VA

Dolloff, C. A., and C. N. Roghair

USDA Forest Service, Southern Research Station, Department of Fisheries and Wildlife Sciences, Virginia Tech, Blacksburg, VA 24061-0321

ABSTRACT — We have studied the basin-wide distribution and relative abundance of brook trout *Salvelinus fontinalis* in Shenandoah National Park's Staunton River (Figure 1) annually since 1993. Additionally, in 1997 we began using a combination of PIT tag recoveries and mark-recapture techniques to examine brook trout movement and growth following a 500+ year flood and debris flow. Our analysis illustrates high variation in year-class strength and population density (Figures 2 and 3), changes in fish growth rates associated with the debris flow, and movement potential of brook trout (Figure 4). The Staunton River population is resilient to acute, stochastic events such as floods, though there can be long-lasting effects on attributes such as growth rate. Brook trout are capable of relatively long-distance movements, though only a small portion of the population undertakes such journeys. Lessons learned over the past 17 years guide us as we adapt the current study to further examine movement as related to fine scale measurements of stream habitat and the potential role of climate change on resiliency of the brook trout population.

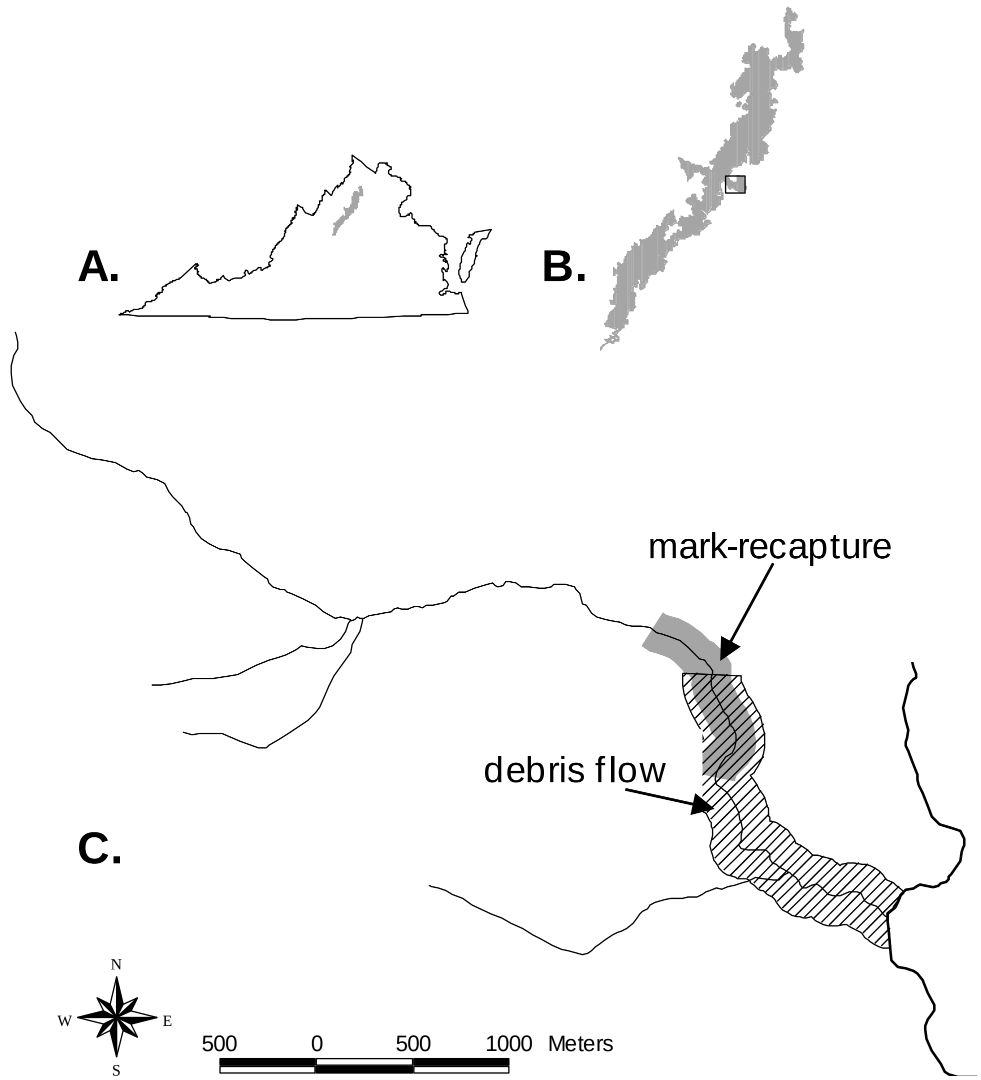


Figure 1. Location of (a) Shenandoah National Park (SNP), (b)Staunton River within SNP, and (c) the Staunton River with the debris flow affected area and mark-recapture study section.

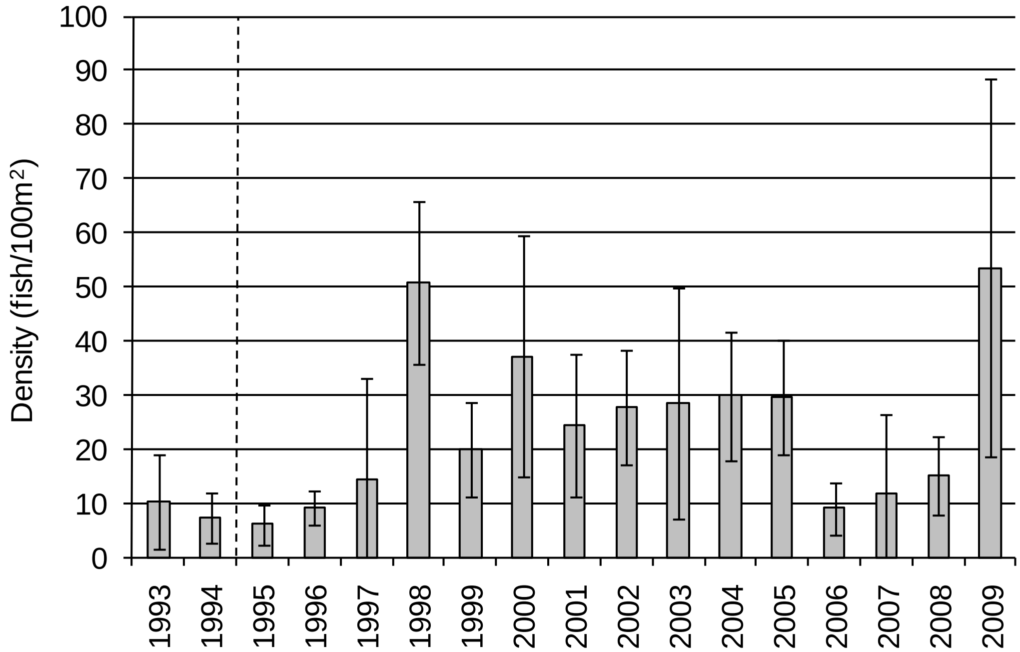


Figure 2. Density of adult brook trout in Staunton River pool habitat as estimated using basin-wide visual estimation technique surveys. Vertical dashed line indicates date of debris flow.

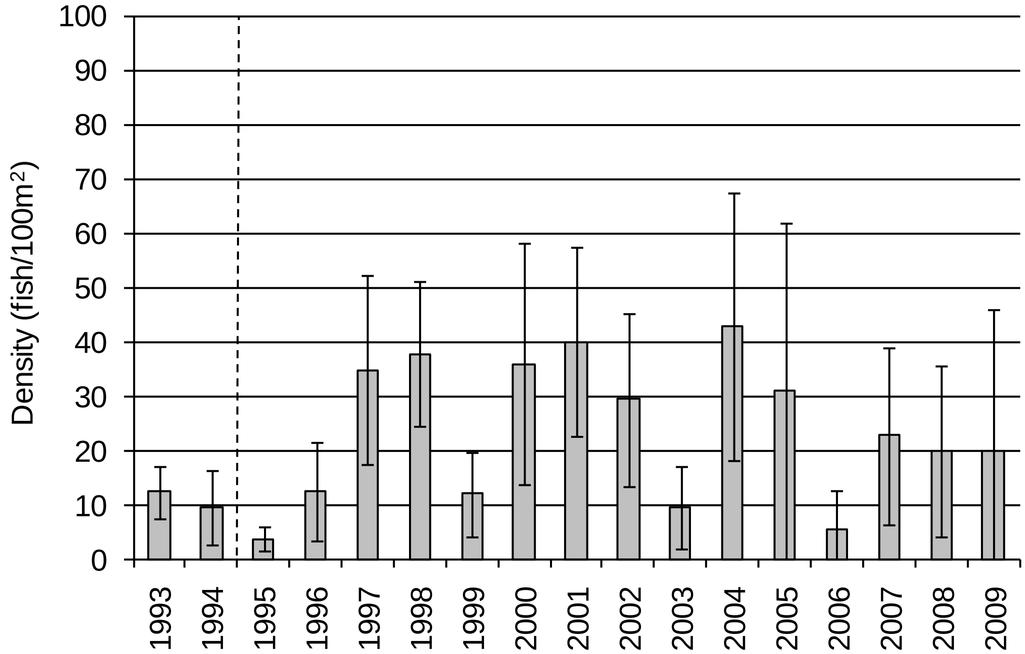


Figure 3. Density of young-of-year brook trout in Staunton River pool habitat as estimated using basin-wide visual estimation technique surveys. Vertical dashed line indicates date of debris flow.

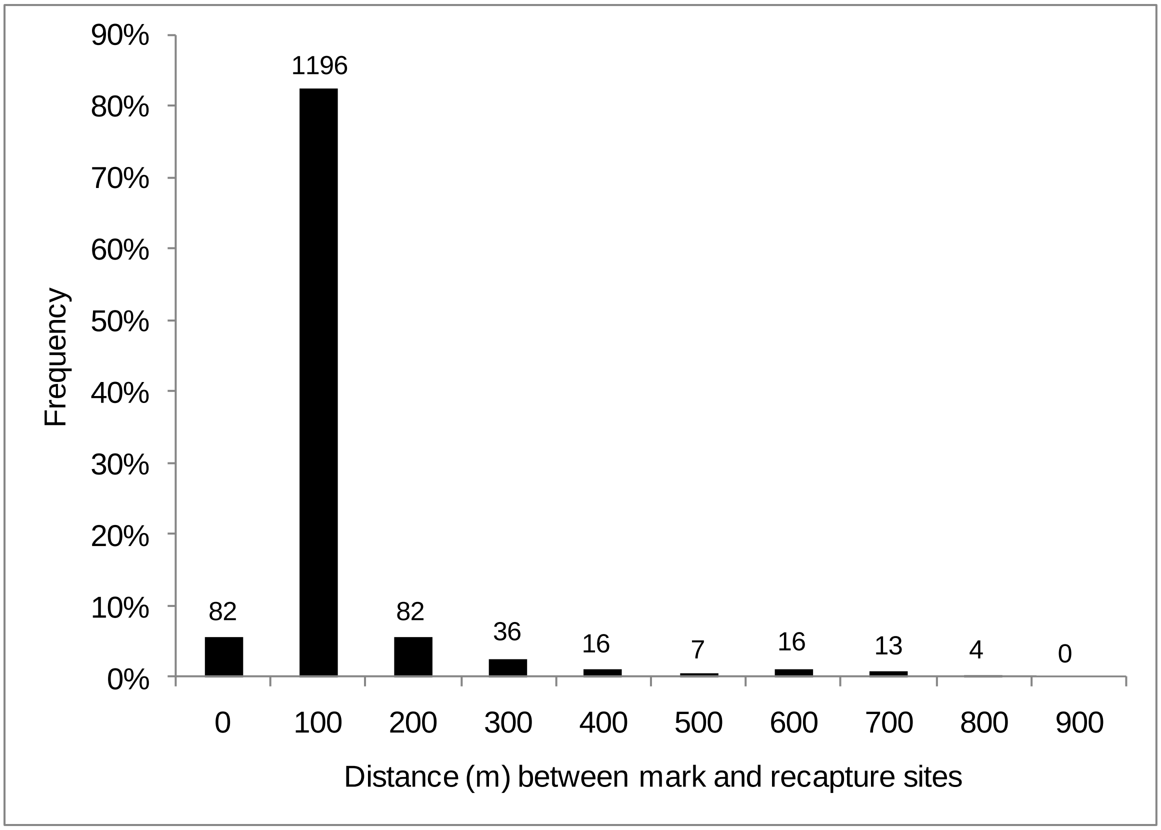


Figure 4. Distance between sites of marking and recapture of fish implanted with PIT tags within a 970-m long reach of the Staunton River between 1997 and 2010; n = 1452.

CALIFORNIA GOLDEN TROUT AND CLIMATE CHANGE: IS THEIR STREAM HABITAT VULNERABLE TO CLIMATE WARMING?

Kathleen R. Matthews

Research Scientist USDA Pacific Southwest Research Station PO Box 245 Berkeley, CA 94701. 510-559-6454; 510-559-6440; kmatthews@fs.fed.us

ABSTRACT — The California golden trout (CGT) *Oncorhynchus mykiss aguabonita* is one of the few native high-elevation fish in the Sierra Nevada. They are already in trouble because of exotic trout, genetic introgression, and degraded habitat, and now face further stress from climate warming. Their native habitat on the Kern Plateau meadows mostly in the Golden Trout Wilderness (GTW) currently includes stream areas impacted by cattle grazing. As a result, some areas have reduced streamside vegetation (willows or sedge) and widened channels with shallow stream depths that often lead to warmer water temperatures. Climate change will further compromise CGT and their habitat in stream areas still being grazed, because the warmer water temperatures predicted under most warming scenarios could increase water temperatures to lethal levels. One important management response to climate warming will be to ensure that habitats are more resilient to predicted changes in water temperature, flow, and snow pack. I have initiated a study to determine the climate change resiliency of golden trout habitat by conducting a spatially explicit analysis of stream temperatures in restored and degraded sections of meadows in the GTW. Preliminary data from 2008 to 2010 indicate that stream temperatures are already 24^{o}C in degraded areas. The detailed temperature profiles will also be used to estimate what proportion of the habitat will be resilient to climate change and what proportion should undergo increased restoration. Because water temperatures are already approaching stressful limits, management action to restore CGT habitat is imperative.

INTRODUCTION

The California golden trout *Oncorhynchys mykiss aguabonita (*Figure 1*)* is native to the South Fork Kern River and Golden Trout Creek (Behnke 1992), and most of its native range lies within the GTW (Figure 2). Stream populations in the GTW are long lived, slow growing, and exist at high densities because of increased spawning habitat due to cattle grazing (Knapp and Dudley 1990; Knapp and Matthews 1996; Knapp et al. 1998). The California golden trout (CGT) has been the subject of management interest because of its status as California's state fish, its limited natural distribution, and several perceived threats to its viability, including introduction of nonnative brown trout *Salmo trutta* and habitat degradation caused by livestock grazing. Because of these threats, the California golden trout is being considered for federal listing as a threatened species. The CGT was the focus of a major restoration effort in the 1970s-1980s (Pister 2008) to remove exotic trout throughout its native range. Several studies have described golden trout habitat preferences (Matthews 1996a, 1996b) and concluded that adult trout prefer habitats (vegetated and undercut banks) typically reduced by cattle grazing. One emerging threat that has not been evaluated is the impact of climate warming that could further stress CGT streams already near their temperature maximums due to cattle grazing.

METHODS

To determine the water temperature vulnerability of the CGT within its stream habitat in the GTW, I deployed temperature probes (Onset HOBO Water Temp Pro v2) throughout the stream to include a typical range of stream areas: areas degraded by cattle grazing (collapsed banks, little vegetation, shallow depth) and those in recovering areas with more vegetation and greater depth. Potentially harmful water temperatures will be reported to managers so that actions can be taken. The analysis is being conducted in Mulkey, Ramshaw, and Big Whitney meadows in the GTW (Figure 2). The detailed temperature and dissolved oxygen (DO) profiles will be used to estimate what proportion of the habitat will be resilient to climate change and what proportion should undergo increased restoration.

Figure 1. California golden trout. Photo by Kathleen Matthews.

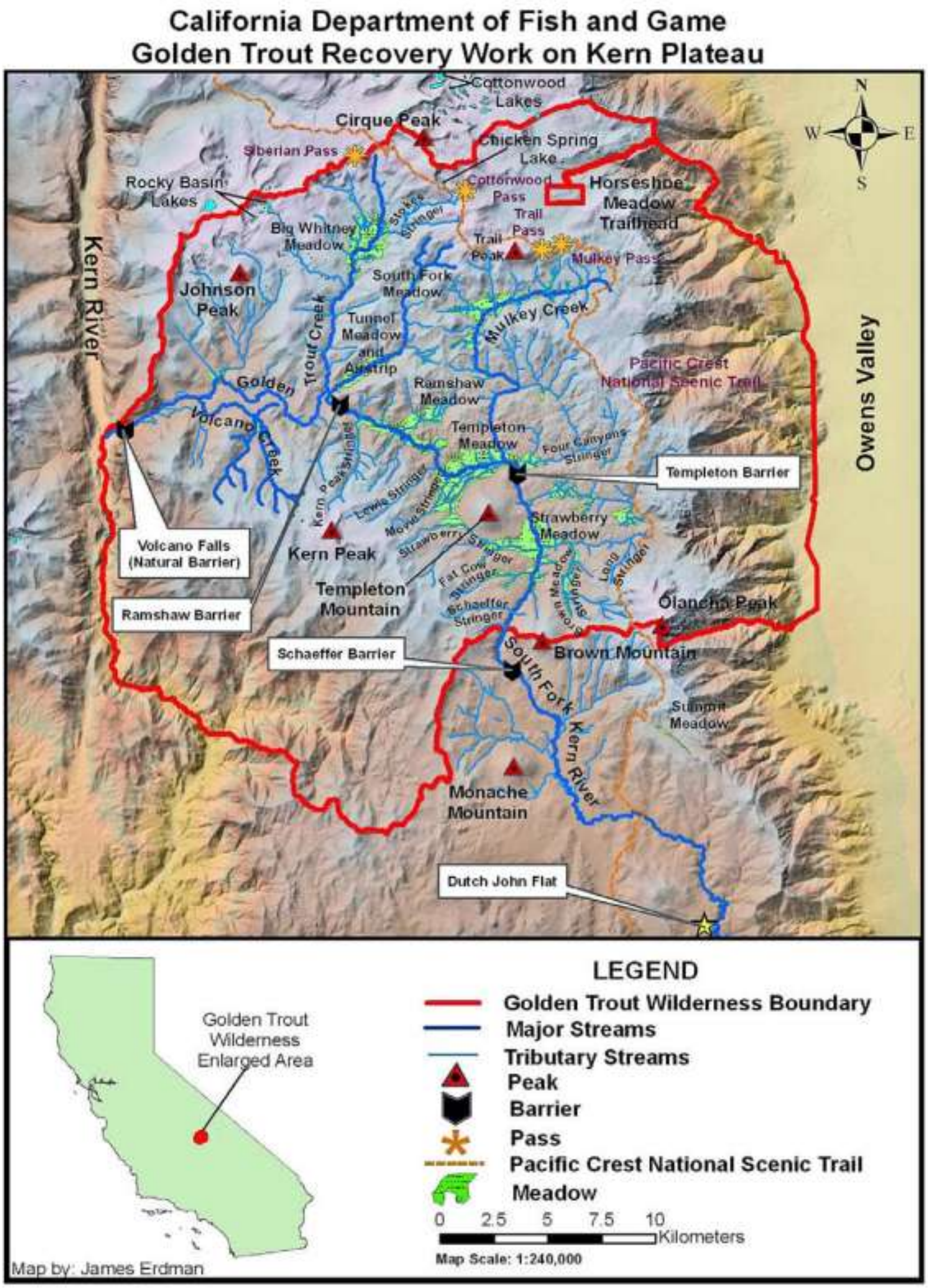


Figure 2. Map of Kern plateau from Pister 2008.

Results

Preliminary temperature data from 2009 show that water temperature reaches 24°C in late July (Figure 3a) in degraded portions of Mulkey Creek (Figures 3a-b). A 24-hour breakdown of temperatures also indicates a 15°C diel change in water temperature (Figure 3 b). In contrast, in recovering sections of Mulkey and Ramshaw meadows (Figure 3c), water temperature did not exceed 22°C and diel changes were 11°C or less.

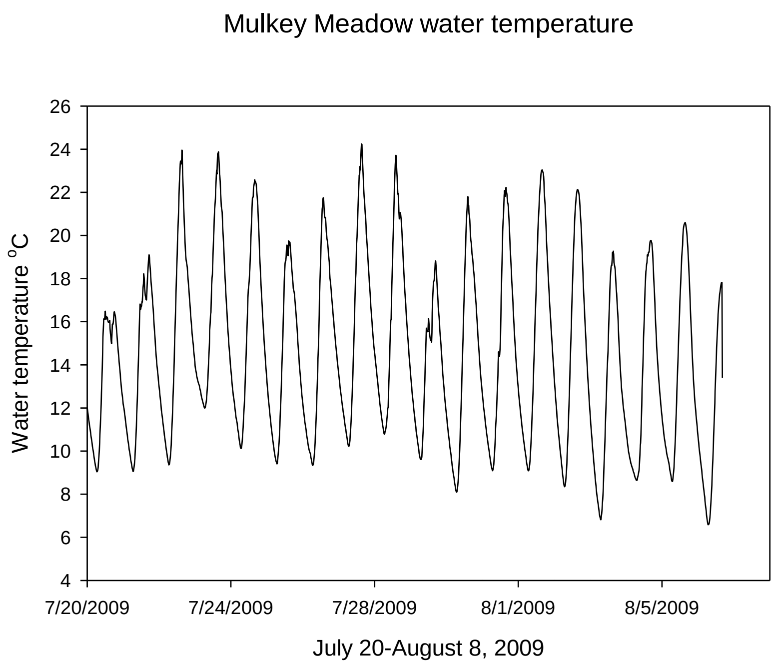


Figure 3a. Water temperatures in Mulkey Creek, Golden Trout Wilderness from July 19-August 8, 2009.

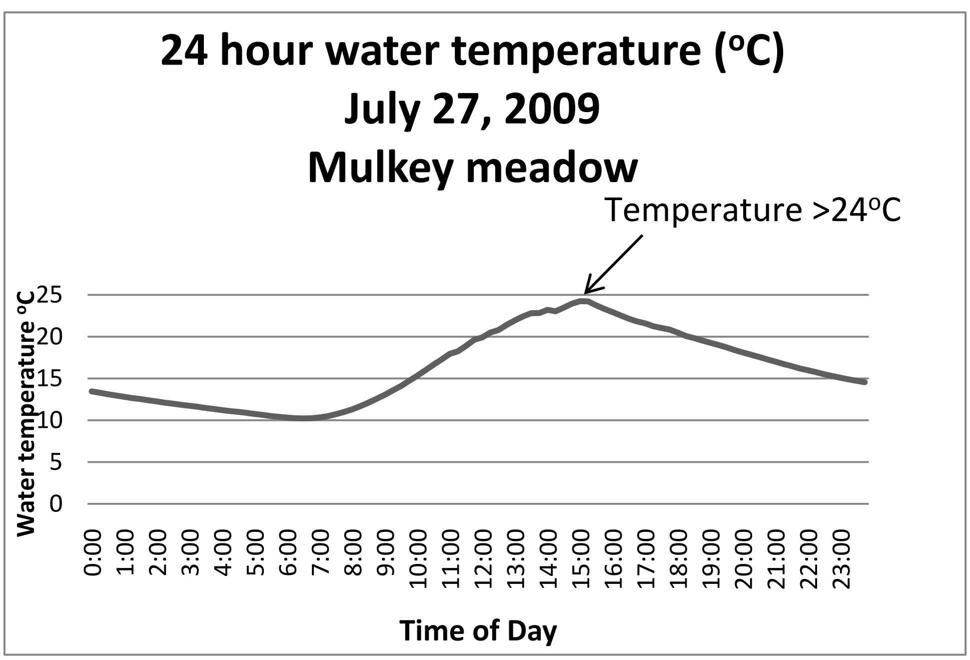


Figure 3b. Diel change in water temperature on July 27, 2009.

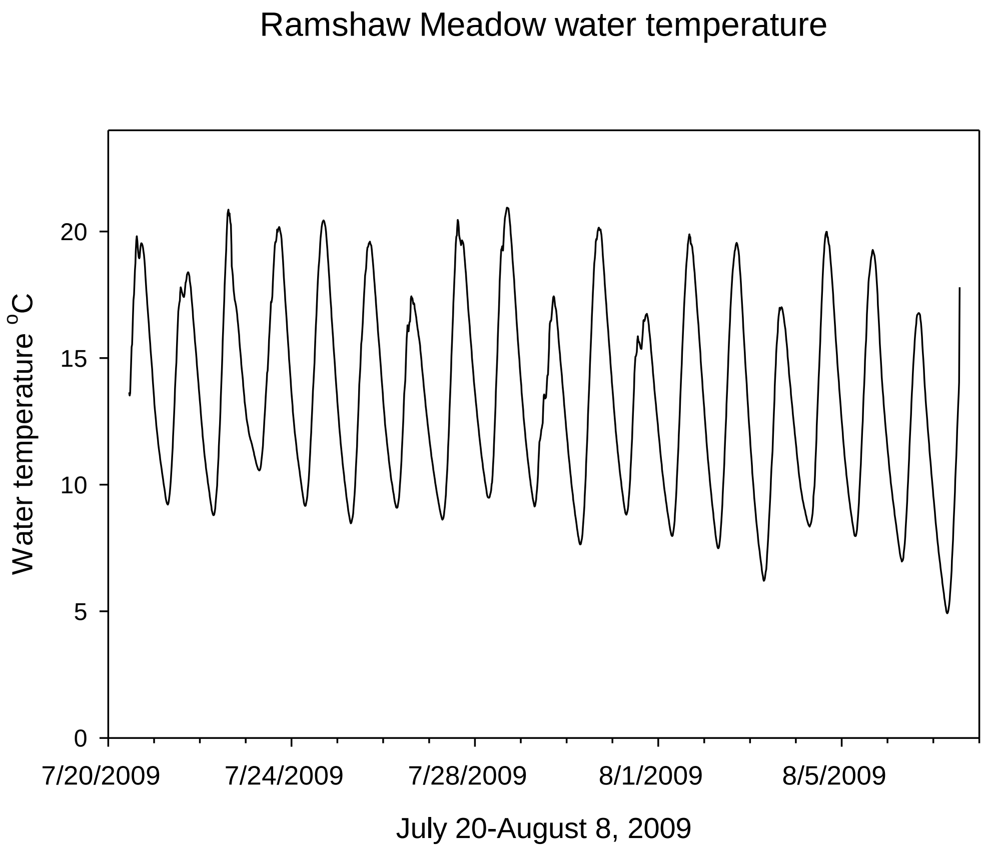


Figure 3c. Water temperatures in Ramshaw Meadow South Fork Kern River, Golden Trout Wilderness from July 20 to August 8, 2009.

Figures 4a and 4b. Sections of Mulkey Creek experiencing highest water temperatures.

Figure 5. Restored area within Ramshaw Meadow stream with maximum water temperatures never exceeding 22°C.

Discussion

Preliminary water temperature data from 2009 shows some vulnerabilities: water temperatures in some degraded areas of the GTW are already experiencing stressful levels (>24°C) (Bjorn and Reiser 1991), and these areas will not be resilient to future climate warming. Future climate warming could increase water temperatures to lethal levels stressing the importance of continued monitoring. Moreover, the diel range of temperatures (± 15°C) experienced by CGT is extreme (Matthews and Berg 1997) and warrants further study because it may physiologically stress trout. Water temperatures will be monitored through 2011.

In the Sierra Nevada Mountains (California, USA), there is great opportunity to increase resiliency of high elevation aquatic habitats because most of it is within federally designated Wilderness set aside by U.S. Congress to "to preserve its natural conditions and which generally appears to have been affected primarily by forces of nature" (Kloepfer et al. 1994). Indeed, most of the CGT habitat is within the Golden Trout Wilderness; hence, there is the opportunity to reduce or eliminate activities that reduce the resiliency to increased climate warming. While Wilderness status typically prohibits logging, roads, and mechanized equipment, it does allow cattle grazing, that often impacts stream habitat and increases water temperatures. To provide more resiliency of important habitat for aquatic species, Wilderness areas could be used as refuges, i.e., the freshwater version of marine preserves. In these preserves, managers could eliminate or minimize activities that are currently allowed, such as cattle grazing, but are lowering the resiliency of freshwater habitats to increased warming.

References

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society Monograph 6, Bethesda, Maryland.

Bjornn, T. C., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. Pages 83-138 *In* W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19. Bethesda, Maryland.

Kloepfer, D., J. Watson, and P. Byrnes. 1994. The Wilderness Act Handbook. The Wilderness Society. Washington. D.C.

Knapp, R. A., and T. L. Dudley. 1990. Growth and longevity of golden trout, *Oncorhynchus aguabo-*

nita, in their native streams. California Fish and Game 76:161-173.

Knapp, R. A., V. T. Vredenburg, and K.R. Matthews. 1998. Effects of stream channel morphology on golden trout spawning habitat and population structure. Ecological Applications 1104-1117.

Knapp, R. A. and K. R. Matthews. 1996. Livestock grazing, golden trout, and streams in the Golden Trout Wilderness, California: impacts and management implications. North American Journal of Fisheries Management 16:805-820.

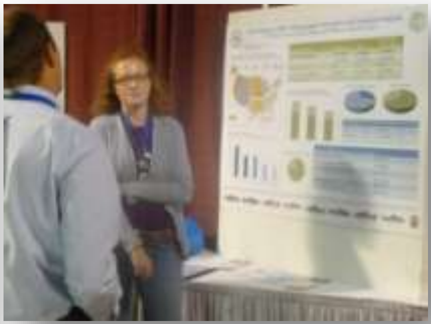
Matthews, K.R. 1996a. Golden trout habitat selection and movement patterns in degraded and recovering sites within the Golden Trout Wilderness, California. North American Journal of Fisheries Management 16:578-590.

Matthews, K. R. 1996b. Diel movement and habitat use of the golden trout *Oncorhynchus mykiss aguabonita* in their native habitat on the Golden Trout Wilderness, California. Transactions of the American Fisheries Society 125:78-86.

Matthews, K. R. and N. H. Berg. 1997. Rainbow trout responses to water temperature and dissolved oxygen stress in two southern California stream pools. Journal of Fish Biology 49:1-18.

Pister, E. P. 2008. Restoration of the California golden trout in the South Fork Kern River, Kern plateau, Tulare County, California, 1966-2004, with reference to golden trout creek. California Department of Fish and Game Administrative Report No. 2008-1, Sacramento.

Session 2: Economic and Cultural Values of Wild Trout



THE CHALLENGING ROLE OF ECONOMICS IN THE DESIGNATION OF CRITICAL HABITAT

Edward J. Maillett

Senior Economist, U.S. Fish and Wildlife Service, Division of Economics, Arlington, Virginia. Contact Information: edward_maillett@fws.gov; 703-358-2322.

ABSTRACT — This paper provides a brief overview of some of the challenges facing the U.S. Fish and Wildlife Service in incorporating economic information as part of its critical habitat designation process. The Service is required to explicitly consider the economic impacts of a designation before making any final determinations. As the critical habitat designations have become more numerous and in many cases, encompass large amounts of area involving many diverse groups of stakeholders, the framework and scope of the economic analysis has undergone considerable scrutiny. Not only is the analysis meant to aid decision-making, but it is also unique in that it describes the interpretation of the rule and its effects on the general public. The designation of critical habitat for the bull trout *Salvelinus confluentus* provides a good example to highlight some of these issues as it is also one of the few designations that expressly factored economic impacts into its final exclusions.

INTRODUCTION

The Endangered Species Act (Act) requires critical habitat to be designated when a species is afforded protection under the Act at the time it is listed. Critical habitat is defined under the Act to be those areas considered essential for the conservation of the protected species.[1] While the Act stipulates that the decision-making process to list a species be based on science, the decision-making process for the designation of critical habitat is required to also consider the economic impacts before making any final determinations. This paper will provide an overview of some of the challenges facing the U.S. Fish and Wildlife Service (Service) in incorporating economic information into its critical habitat rulemakings and in doing so, will highlight the case history of the critical habitat designations for the bull trout *Salvelinus confluentus* as it pertains to the continuing evolution of the economic analysis framework.

The Service initially proposed to designate critical habitat for the Columbia and Klamath River distinct population segments of the bull trout in November 2002, after listing the species as threatened under the Act in June 1998. In June 2004, the Service proposed to designate critical habitat for the Jarbidge, Coastal-Puget Sound, and Saint Mary-Belly populations after listing these species as threatened in 1999. In October 2004, the Service published a final critical habitat rule for the Columbia and Klamath populations. This final rule was one of the first to cite economic concerns as a rationale for making certain exclusions. The rationale behind these exclusions did not last long, however. In September 2005, the Service issued a final critical habitat rule for the Jarbidge, Coastal-Puget Sound, and Saint Mary-Belly populations and at that time also revised their final critical habitat determinations for the Columbia and Klamath populations. While many of the areas originally excluded under the original final critical habitat rule remained, the Service no longer cited economic concerns as the explicit rationale for exclusion.

While the Act has permitted decision-makers to exclude areas from critical habitat for economic impacts since it was amended in 1978 (partly in response to the Tellico Dam controversy) in reality this provision has been rarely exercised. Beginning in 2001, however, when the flood of lawsuits began demanding the Service to designate critical habitat for listed species, a new Administration came to

[1] The term "conservation" is defined under the Act to mean the use of all methods and procedures necessary to recover the species and remove it from its protected status under the Act.

govern that was particularly interested in understanding who and how these designations would impact landowners. This raised the status of the economic analysis, which in turn raised a number of issues regarding how impacts were to be assessed and presented in a document that was increasingly playing a more direct role in the decision-making process. Among some of the issues that were raised at that time, which are still being discussed today are

1. The uniformity of proposed essential habitat areas;
2. The spatial extent of an "area" in the economic analysis;
3. The identification and treatment of baseline conservation measures; and
4. How best to express potential economic benefits.

Each issue is discussed in greater detail, below.

Uniformity of Essential Habitat

Critical habitat is to be only those areas considered to be *essential* for the conservation of the species. However, this creates an automatic conflict with the mandate to consider the effects of economic impacts before making any final determinations. If the habitat is truly essential for conserving the species, then it stands to reason that it cannot be excluded from any critical habitat designation for economic reasons or else the goal of conservation could not be attained. Even after a final determination is made, however, it is not uncommon to see permitted land-use modifications suggesting that at least on a marginal scale certain areas of critical habitat may in fact be less essential for conservation than others.

In many cases, including the bull trout, the proposed critical habitat areas are so large that the Service cannot reasonably be expected to ascertain the uniformity of the entire area in terms of its habitat values for both the survival and recovery of the species. Instead, the Service defines a set of habitat characteristics (primary constituent elements) to clarify what landowners need to look for once their land has been officially included within the boundaries of a designation to determine whether or not their land has any of the physical features necessary to make it truly critical habitat for the listed species.

Most designations, including the bull trout, identify multiple habitat features that are considered to be essential for the conservation of the species. For the Columbia and Klamath populations of the bull trout, these features include (1) space for individual and population growth; (2) food, water, or other nutritional requirements; (3) shelter; (4) breeding, reproduction, and rearing sites; and (5) habitats that are protected from disturbance. Clearly not all of these features will be uniformly present across every single area and clearly some of the areas will have better quality features than other areas. Over time this becomes more evident through the collection of more detailed data.

For example, the Service initially proposed designating 20,980 stream miles (18,449 miles was considered to be occupied) and 591,577 acres of lakes and reservoirs (561,481 acres were considered occupied) for all five populations of the bull trout in their 2002 and 2004 proposed rules. Later, in 2010, the Service changed its proposal for these populations to include 22,679 stream miles (21,718 occupied miles) and 533,426 acres of lakes and reservoirs (517,550 occupied acres). In both instances, the proposals were for areas considered to be essential for the conservation of the bull trout. The 2010 proposed rule explains the discrepancies to be based on better occupancy data and refined information on the importance of certain habitats. While the differences appear to be small in context of the entire amount of area being considered for designation, it illustrates the dynamics associated with defining essential areas. Developing a method to account for informational uncertainties in a quantitative manner combined with an accounting of the relative contribution of particular areas in terms of both current and future contributions for a species survival and recovery could aid the conservation process.

In fact, the Service's sister agency, the National Marine Fisheries Service (NMFS) formerly recognized the habitat variability in their critical habitat designations for the Pacific *Oncorhynchus spp.* and Atlantic salmon *Salmo salar*. For these two rulemakings, NMFS developed a process to score areas proposed for critical habitat based on such biological criteria as the quantity and quality of the spawning and rearing characteristics of the habitat along with the migratory needs of the species. Habitat areas were given a ranking of either high, medium, or low based on their relative conservation value to the species. Also, NMFS considered the probability that an area would be subject to a future consultation under the Act. This information was then weighed against the economic impacts associated with consultations under the Act for each area. The final critical habitat designation for the Pacific salmon and steelhead *O. mykiss* contained multiple economic exclusions, which were determined following a two-step process. Under the first step, NMFS identified all areas eligible for exclusion based on a relative scale of economic impact. Next, biological teams were asked to consider whether excluding any of the eligible areas, either alone or in combination with other eligible areas, would significantly impede conservation. Areas identified as high economic impact having low conservation value were excluded from the final designation thus avoiding potentially wasteful conservation measures in relatively unproductive areas.

Spatial Extent of an "Area"

The second issue that the Service has wrestled with over the years pertains to the spatial extent of the economic analysis. The Act is clear in that before making a final determination about a critical habitat designation, the Service must take into consideration the economic impact of specifying *any particular area* as critical habitat. Areas can be excluded from critical habitat if it is determined that the benefits of exclusion outweigh the benefits of inclusion, unless the exclusion would result in the extinction of the species.

While the Act allows for the exclusion of any particular area if the benefits of exclusion outweigh the benefits of inclusion, the relationship of an area to a critical habitat unit is not defined. In addition, economic data are often only available along sociopolitical boundary lines, which often do not match up with critical habitat boundaries. Thus, the decision regarding the spatial detail of the economic analysis is frequently a balancing act, which must incorporate such factors as the time given to conduct the analysis, spatial extent of the designation, size of individual critical habitat units, and degree of stakeholder interest.

In general, the more refined an economic analysis becomes, the more time and resources it takes to conduct the study and present the findings in a useful format for decision-making. The benefit of such refinement is that it allows for the exclusion of finely targeted, high impact areas. When an economic analysis is conducted at a coarser scale, high impact economic activities may end up being lumped in with surrounding low impact areas, potentially resulting in inefficient economic exclusions, from a conservation perspective.[2]

The bull trout critical habitat rules provide a convenient example for illustrating the different methods used to define an area. In the November 2002 proposed critical habitat rule for the Columbia and Klamath distinct population segments, the Service's proposal was broken into 25 distinct critical habitat units based on geographic location. Economic impacts were assessed at the critical habitat unit level. The analysis found that the Willamette River Basin was forecasted to incur the most economic impact both in terms of total impact per unit as well as on a cost-per-river- mile. The analysis also reported that five of the 25 units in total accounted for over 50% of the total impact and that two units alone accounted for 25% of the total impact on a river-mile basis. Interestingly, these two units (Willamette and Malheur) accounted for just over 2% of the proposed river miles of the designation.

Based on these findings, the 2004 final critical habitat rule excluded both the Willamette and Malheur units specifically because of high economic impacts. This was one of the very first explicit economic exclusions in a critical habitat rulemaking. In addition, the final rule also collectively excluded waters impounded behind dams specifically out of concern for the potential economic impacts that were detailed within each unit. As previously mentioned, the rationale for excluding these areas for economic considerations did not last long. On September 26, 2005 the Service revised its 2004 final rule and dropped any explicit reference to economic impacts as a rationale for excluding any areas without explanation.

During this period, it became apparent that the decision-makers were interested in better understanding economic impacts at a finer scale before

[2] The first critical habitat rule to explicitly exclude proposed areas for economic impacts was for the four vernal pool crustaceans and eleven vernal pool plants in California and southern Oregon in August 2003. Due to several factors, the economic analysis estimated impacts at a county-wide scale. As a result, the final rule excluded several entire counties based on economic impacts although in reality the distribution of impacts within each county was not uniform across the areas proposed as critical habitat.

making any final determinations. The Service similarly supported such an approach to avoid large scale exclusions where the economic impacts may be concentrated in just a small area within a unit. In response, the economic analysis for the proposed critical habitat rule for the Jarbidge, Coastal-Puget Sound, and St. Mary Belly bull trout was conducted at the fifth-field Hydrologic Unit Code (HUC), as defined by the U.S. Geological Survey. This gave the reader a deeper understanding of where exactly particular areas may incur disproportional or significant impacts beyond a simpler assessment conducted at the larger critical habitat unit scale.

For example, the economic analysis determined that the Lower Green critical habitat subunit was the most impacted within the entire Puget Sound unit. However, the analysis went further and identified the specific watershed within this subunit that incurred the most impact, which was the Lower Green watershed (HUC 1711001303) that accounted for 75% of the total subunit's impact. While the analysis was able to identify high cost area HUCs, which would allow a decision-maker to more finely exclude areas without affecting an entire unit or subunit, no exclusions were made based on economic impacts.

Consideration of Existing Conservation Protection Measures

Typically an economic impact analysis assesses the state of change to society based on the expected difference between a "with" and "without" scenario of the proposed rule under consideration. However, for critical habitat rulemakings, following this standard has produced some confusing results. This is because once a species is protected under the Act, even absent the designation of critical habitat, the species along with its habitat is afforded significant protection. Once listed, federal agencies are required to consult with the Service on any action authorized, funded, or carried out to ensure that such action is not likely to jeopardize the continued existence of the listed species. Only after the designation of critical habitat is the Service compelled to consider the effects of an agency action in terms of whether or not it will also adversely modify critical habitat. The two terms, jeopardy and adverse modification were defined in such a similar manner that over time it became practically difficult to distinguish between actions that could jeopardize the species from those destroying or adversely modifying critical habitat.[3]

The nearly identical definitions fostered the Service's position that the conservation benefits afforded listed species through critical habitat designations were extremely marginal, particularly in areas determined by the Service to be occupied by the species. As a result the Service attributed the majority, if not all of the conservation protection measures to the listing process, which is not subject to any consideration of economic impacts. Consequently, many of the earlier economic analyses for critical habitat designations concluded that there would be no additional economic impacts resulting from such a designation.

However, a key event occurred that had a significant effect on how the economic analysis was conducted. On May 11, 2001, the United States Court of Appeals in the Tenth Circuit decided that the critical habitat economic analyses conducted by the Service were inadequate. Specifically, the Court found that an economic analysis that was focused solely on an assessment of impacts that were uniquely attributable to a critical habitat designation was virtually meaningless because the Service had been treating the protections afforded a species' critical habitat as co-extensive with the protections afforded a species' habitat through the listing process. While the Court explicitly recognized that the root of the problem lay with the similarity of the regulatory definitions, it was only able to instruct the Service to conduct a meaningful economic analysis that, if it must, assessed the impacts associated with the avoidance of jeopardizing a species to the extent that such actions would also co-extensively avoid adversely modifying its critical habitat.

This led to a broad expansion of the economic analysis because at the time there was no clear guidance on how to practically distinguish between

---

[3] The term "destruction or adverse modification" was defined at 50 CFR 402.02 as a direct or indirect alteration that appreciably diminishes the value of critical habitat for both the survival and recovery of a listed species, while the term "jeopardize" means the continued existence of means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species.

the two standards during consultation, particularly for occupied habitat. Consequently, both of the bull trout economic analyses reported co-extensive economic impacts associated with the proposed designations. Co-extensive impacts included reasonably foreseeable actions having a federal nexus that may first require consultation with the Service before getting permission to proceed. Both of the analyses also had to wrestle with the fact that already there were significant baseline protection measures that benefited the bull trout and its habitat along with other species. Determining what was attributable to bull trout and what was not was extremely difficult, and the analyses made clear that many of the land and water management practices and standards could be traced to measures designed to emphasize habitat protection for Pacific salmon.

This relates to a second complicating factor. Not only did bull trout habitat overlap in many areas with protected Pacific salmon habitat, NMFS was simultaneously proposing critical habitat for the salmon as well. There was considerable concern, particularly from oversight agencies as well as interested stakeholders, that both analyses could potentially claim as baseline protection those protection measures afforded to the other species as a result of its listing and critical habitat designation. If such a framework were to be followed, both analyses would have failed to properly access the full economic impact associated with the rulemaking and thus run afoul of other federal rulemaking requirements leading to the certainty of future lawsuits.[4] Given the backlog of critical habitat designations needed to be promulgated by the Service at this time and the desire to avoid future litigation particularly in regards to the economic analysis, considerable effort was taken to recognize within the analysis the likelihood of any future overlapping conservation measures with the salmon, but to also include such impacts as co-extensive with the bull trout to the extent that such measures were also necessary to conserve the bull trout as part of a future project should the salmon not have shared in the bull trout's habitat (e.g. construction of fish ladders).

The Columbia and Klamath economic analysis analyzed each of the 25 units for reasonably foreseeable land use management actions having a federal nexus that may require changes in project management or behavior to avoid adversely modifying bull trout critical habitat even if such measures co-extensively avoided jeopardizing the species. Undertaking this task was even more difficult given that the proposed rule did not specifically identify or discuss the types of special management or habitat protection measures that may be necessary to conserve the species. Instead, the proposed rule focused on describing the features necessary for bull trout survival and recovery. In response, the economics team collected as many bull trout section 7 consultations as it could and went through them to determine the types of actions that triggered section 7 consultations, the special management protection measures described in the consultation, the federal action agency, and if applicable, any non-federal third-party associated with the outcome of the consultation.

The draft economic analysis of the Columbia and Klamath proposed rule concluded that the co-extensive conservation-related impacts associated with the designation of critical habitat impacts to range between \$20 million to \$26 million (U.S.) annually and that federal agencies would incur approximately 70 to 75% of the total costs. It found that most of the forecast project modification costs were dam and reservoir related (42%) with other conservation costs associated with timber harvest (29%), USFS-related water diversions (12%), habitat conservation plans (8%) and placer gold mining and other events (3%). The economic analysis for the Puget Sound, Jarbidge, and St. Mary Belly bull trout found that the total co-extensive impact of conservation-related impacts to be \$60.8 million, annually and that the highest economic impact was associated with conservation measures associated with residential and commercial real estate development (\$26.1 million, annualized), which represented about 44% of the total co-extensive impacts. Conservation measures associated with hydroelectric projects were

[4] There are a number of analytical requirements associated with federal rulemakings that are independent of the Endangered Species Act. Executive Order 12866 requires agencies to formally assess both the economic costs and benefits of their regulations (including cumulative effects) and to consider regulatory alternatives that will minimize the burden to regulated entities. The Regulatory Flexibility Act requires agencies to determine whether or not their regulations will impose a significant impact on a substantial number of small entities and if it does, develop a Regulatory Flexibility Analysis containing regulatory alternatives to minimize the burden on small entities.

estimated to be only $5.1 million, annually, or about 8% of the total.

In contrast, the Service recently re-proposed critical habitat for all of the bull trout populations and has modified its definition of baseline conditions. The economic analysis for this proposal attempts to more formerly distinguish between economic impacts directly attributable to a critical habitat designation from those that may occur co-extensively with listing protections. To do this, the Service had to develop an Incremental Effects Memo that detailed to the economists exactly how conservation measures would be applied with and without critical habitat. The memo instructed the economics team to expect no differences in conservation measure outcomes for consultations involving occupied critical habitat but that the administrative process of considering the impacts of a proposed project on critical habitat would add a 33% additional administrative burden to the Service and associated action agencies. The memo stated that only conservation measures associated with areas identified as unoccupied critical habitat were to be attributed to the proposed rule. Accordingly, the draft economic analysis for a rule very similar to the earlier proposals now estimates economic impacts to range between $5.0 - $7.1 million per year. Still, the greatest impact is expected to be associated with dam modification projects, such as the installation of fish passages, temperature controls, and flow monitoring and management for the species.

Economic Benefits

For the 2002 proposed rule, a preliminary estimate of the economic benefits was conducted as part of the initial draft report on economic impacts. Economic benefits were broken down into four distinct categories: (1) direct (use) benefits; (2) existence values; (3) indirect benefits (i.e., ecosystem services); and (4) total value. Only direct-use benefits were estimated, while the other types of benefits were described anecdotally.

The draft chapter estimated that the direct benefit associated with a restored bull trout sport fishery to be about $6 million per year or less in the Columbia Basin and $100,000 per year or less in the Klamath Basin. These estimates were based on the assumptions that a restored Klamath bull trout fishery would result in an additional 3,000 to 4,000 days per year of bull trout fishing in streams and that a restored Columbia Basin bull trout fishery would result in an increase between 218,000 to 269,000 angler days each year and that the economic surpluses associated with a restored bull trout fishery ranged between $17 per day for in-state anglers to $50 per day for out-of-state anglers.

One issue that was raised internally in reviewing this estimate was with the assumption made in the economic analysis that critical habitat would lead to a fully recovered bull trout population that could be freely targeted by sportsmen within 25 years. Although Service biologists certainly supported this goal, at the time they could not necessarily agree that the designation of critical habitat would lead to recovery within the next 25 years. To the extent that a delisting occurs further out in time, the present value of the future stream of economic benefits would be lower than that reported. Also, to the extent that a restored fishery results in fewer additional angler days than forecasted (due to demographic changes in preferences over time, for example), the economic surplus and increased angler day estimates would also be overstated.

The second component of economic benefit, existence value, relates to the concept that certain members of our society place a value on simply knowing that an endangered or threatened species continues to exist in its natural environment and are willing to pay to support this benefit. The analysis surveyed the economics literature for published studies that estimated the existence values for other endangered and threatened fishes. None of these studies related to the bull trout and given the fact that the reported values in the studies varied widely depending on the species, location, and survey method, made the authors reluctant to attempt any type of credible benefit-transfer method.

Another potential economic benefit discussed in the draft was the potential for indirect benefits. Indirect economic benefits could include project modification cost savings for other listed species that concurrently benefit from bull trout conservation measures, improvements or avoidance of degradation of certain ecosystem services (e.g., drinking water), and benefits to certain types of recreationists through the maintenance of in-stream flows. Many of these benefits potentially overlap one another, making estimation difficult to credibly quantify. In addition, as previously mentioned, NMFS was in the process of promulgating their own set of critical habitat rules for the Pacific salmon, which further complicated the reporting of impacts without double counting.

The chapter also discussed an alternative perspective for assessing the beneficial economic impacts associated with the conservation of the bull trout – total value. Total value in this context refers to the value placed on all possible motivations or uses including direct and indirect use, and existence motives. Total value may be reflected in the resources society chooses to invest in such conservation actions as fish and wildlife mitigation actions under the Northwest Power Act and the decision by the Confederated Salish and Kootenai Tribes to implement its Wetland/Riparian Habitat and Bull Trout Restoration Plan.

Conclusions and Recommendations

Below are several suggestions aimed to improve the efficacy of the economic analysis as well as to clarify its role in the decision-making process based on some of the examples discussed.

First, adopting new and clearly distinct regulatory definitions for the terms jeopardy and adverse modification would enable all stakeholders under the critical habitat process to better understand the differences in conservation standards and expectations with and without critical habitat. This will aid conservation by making the consultation process under the Act more efficient as stakeholders could better understand and estimate the type and scale of conservation measures that would likely be imposed for their actions without first having to undergo a formal or informal consultation process. This will also help the Service become even more efficient at streamlining the consultation process as it would hopefully result in a reduction in the time and effort involved in a consultation as project proponents would know ahead of time what would be reasonably expected of them for their proposed actions. Finally, new definitions could also help reduce the seemingly endless rounds of litigation pertaining to the scale and scope of critical habitat designations as it would become increasingly transparent to all parties how the designations proposed by the Service will result in actual conservation to our trust species.

Second, in order for the economic analysis to have any real, practical role in the decision-making process, there should be formal recognition that not all areas proposed for critical habitat are uniform in their habitat qualities. Formally grading or ranking areas based on a selected set of habitat qualities and abundances that are unique to each designation would allow for a better understanding of the relative contribution of proposed areas for a species' survival and recovery. Should the Service be given more time and resources to adopt such a framework, the conservation process could become more efficient and productive as limited conservation resources could be better targeted to the areas that would provide the greatest conservation benefit to our trust species. Obviously, the dividends associated with this framework increase the more refined an area is defined. Developing a habitat ranking scale based on the critical habitat units proposed for the Columbia and Klamath populations of bull trout in their 2002 proposed rule would not have been as conducive for identifying areas that are most cost-effective or least cost-effective for conservation as developing a habitat ranking scale based on individual HUCs. Obviously this takes more up-front time and resources but in the long run could provide a very beneficial roadmap for cost-effective conservation.

Third, caution needs to be exercised in the attempt to measure economic benefits from associated conservation actions. In many cases, in contrast to the bull trout, there are likely no foreseeable direct use benefits. Also, out of the nearly 1,400 species protected under the Act, many are unknown to the general population, raising questions about the plausibility of any meaningful existence values. To advocate for a change in policy that measures economic benefits potentially turns the economic impact analysis into a cost-benefit analysis, which would make it even more difficult to defend any decisions not to exclude areas based on disproportional economic effects. The Act requires only that economic impacts be considered by decision-makers. Economic impacts are commonly defined as the net changes in economic activity within a regional economy associated with the government action. In contrast, economic benefits refer a different concept – total social welfare – and include both market and non-market values. There really is no need to formally assess the net change in total social welfare in order to achieve the desired goals and objectives of the Act if instead the decision-making process focuses on following a cost-effectiveness approach based on economic impacts and how species will physically benefit from land-use management changes.

Acknowledgements

The opinions presented in this paper are solely those of the author and do not necessarily represent those of the U.S. Fish and Wildlife Service. The author wishes to thank Dr. Joseph Charbonneau, Division Chief of Economics in the U.S. Fish and Wildlife Service and Dr. Peter Grigelis, Economist in the Office of Policy Analysis of the Department of Interior for many helpful thoughts and suggestions in review of this manuscript. Any errors or omissions are the sole responsibility of the author.

References

Bioeconomics Inc., Potential Benefits of Proposed Critical Habitat, Draft Report, March 2003.

Bioeconomics Inc., Final Economic Analysis of Critical Habitat Designation for the Bull Trout (Columbia/Klamath Populations). September 2004.

Endangered Species Act, 16 U.S.C. §601 et seq. (1973)

Executive Order 12866, Regulatory Planning and Review, September 30, 1993, (58 FR 51735)

Industrial Economics Inc., Economic Analysis of Critical Habitat Designation for the Bull Trout, December 2009.

National Marine Fisheries Service, Designation of Critical Habitat for West Coast Salmon and Steelhead, Final 4(b)(2) Report, August 2005.

National Marine Fisheries Service, Designation of Critical Habitat for Atlantic Salmon (*Salmo salar*) in the Gulf of Maine Distinct Population Segment, Final ESA Section 4(b)(2) Report, 2009.

National Oceanic and Atmospheric Administration, Designation of Critical Habitat for 12 Evolutionarily Significant Units of West Coast Salmon and Steelhead in Washington, Oregon, and Idaho, September 2, 2005, (70 FR 52630)

New Mexico Cattle Growers Assn v. United States Fish and Wildlife Service, 248 F.3d 1277 (10th Cir. 2001).

Northwest Economic Associates (NEA), Economic Analysis of Critical Habitat for Three Populations of Bull Trout - Coastal Puget Sound, Jarbidge River, Saint Mary-Belly River, 2005.

Regulatory Flexibility Act, 5 U.S.C. § 601 et seq.

U.S. Fish and Wildlife Service, Economic Analysis of Critical Habitat Designation for the Northern Spotted Owl, January 1992.

U.S. Fish and Wildlife Service, Proposed Designation of Critical Habitat for the Klamath River and Columbia River Distinct Population Segments of Bull Trout, November 29, 2002, (67 FR 71236);

U.S. Fish and Wildlife Service, Final Designation of Critical Habitat for Four Vernal Pool Crustaceans and Eleven Vernal Pool Plants in California and Southern Oregon, August 6, 2003, (68 FR 46684);

U.S. Fish and Wildlife Service, Proposed Designation of Critical Habitat for the Jarbidge River, Coastal-Puget Sound, and Saint Mary-Belly River Populations of Bull Trout, June 25, 2004, (69 FR 35768);

U.S. Fish and Wildlife Service, Designation of Critical Habitat for the Klamath River and Columbia River Populations of Bull Trout, October 6, 2004 (69 FR 59996);

U.S. Fish and Wildlife Service, Designation of Critical Habitat for the Bull Trout, September 26, 2005, (70 FR 56212)

U.S. Fish and Wildlife Service, "Incremental Effects Memo for Proposed Bull Trout Critical Habitat" emailed to IEc on September 24, 2009.

U.S. Fish and Wildlife Service, Revised Designation of Critical Habitat for Bull Trout in the Coterminous United States, January 14, 2010, (75 FR 2270).

Watson, Philip, et. al., Determining Economic Contributions and Impacts: What is the difference and why do we care? The Journal of Regional Analysis and Policy, 37(2):140, 2007.

DIVERSE VALUES AND RELEVANCE OF NATIVE SPECIES IN WILDLIFE MANAGEMENT PERSPECTIVES FOR WILDLIFE MANAGERS

Mark Gamblin
Idaho Department of Fish and Game, 1345 Barton Rd., Pocatello, ID 83204

ABSTRACT — Native species, including wild trout, are increasingly important in North American public wildlife resource management policy, programs, and resource allocation decisions. Wild trout management policy and programs should reflect the values and expectations of the society we serve. In practice, that responsibility is a challenge for state wildlife management agencies as the demand for natural resources increases and our society evolves from its agrarian-utilitarian roots. Maintaining relevance of wildlife management policy and programs in our evolving society will be essential for the continued success of fisheries and wildlife management in North America.

I suggest that the continued success of North American wildlife management in general and native species in particular will rely on making the objectives of wildlife conservation and stewardship relevant to American society. That relevance will require that conservation objectives are clearly explained to be in the best interest of the individual.

INTRODUCTION

Original ideas and concepts in the field of wildlife conservation are rare indeed. This essay is no exception. Others have sounded variations of this theme for years. I suggest that fish and wildlife conservation fundamentally depends on awareness and acceptance of and support for those principles by the rank and file citizenry of our society. Further, that support becomes more critical as our own population and demand for natural resources grow. As citizens and as public servants, we and our elected or appointed leaders will make increasingly difficult choices that affect the quality and nature of our natural resources and our future opportunities to use and enjoy those resources. The future of fishing, hunting, wildlife viewing and fundamentally, conservation and management of our national wildlife heritage will be shaped in part by how we as conservationists articulate these complex issues. Our message must be coherent and relevant with a rationale that the average American citizen can understand and relate.

The success of the conservation of North American wildlife resources and the conservation of wildlife traditions (hunting, fishing, viewing, hiking, camping) that depend on public access to our trust resources will depend on how well the North American public (United States, Canada, Mexico) under-understands the unavoidable choices that will affect abundance and cost of energy, food, jobs and other "standards of living", including abundant wildlife, open space, clean water, and clear air.

RESOURCE MANAGEMENT CHALLENGES

More than ever, native species are high priorities for wildlife management planning and allocation of public agency resources. As a heritage of current and future generations, highly valued by our society, native species define the landscape we live in, limit our uses of that landscape, and are at times a lightning rod for conflict and controversy. The Endangered Species Act (ESA) is perhaps the best example of our societal priorities and conflicts as it presents vexing challenges to North American wildlife managers. A review of listed and candidate species in Idaho alone illustrates the challenges to wildlife managers and to society (Table 1). Each listed native species brings conflicts and choices that resource managers and public leaders must deal with.

Table 1. ESA Listed, Candidate and Proposed Species in Idaho, 2010.

- **White Sturgeon (Kootenai River), *Acipenser transontanus* (Endangered)**
- **Steelhead (Snake River runs), *Oncorhynchus mykiss* (Threatened)**
- **Sockeye Salmon (Snake River runs), *Oncorhynchus nerka* (Endangered)**
- **Chinook Salmon (Snake River runs), *Oncorhynchus tshawytscha* (Threatened)**
- **Bull Trout, *Salvelinus confluentus* (Threatened)**
- **Columbia Spotted Frog (Great Basin population), *Rana luteiventris* (Candidate)**
- **Greater sage-grouse, *Centrocercus urophasianus* (Candidate)**
- **Yellow-billed Cuckoo, *Coccyzus americanus* (Candidate)**
- **Northern Idaho Ground Squirrel, *Spermophilus brunneus brunneus* (Threatened)**
- **Southern Idaho Ground Squirrel, *Spermophilus brunneus endemicus* (Candidate)**
- **Gray Wolf, *Canis lupus* (Endangered)**
- **Grizzly Bear, *Ursus arctos* (Threatened)**
- **Canada Lynx, *Lynx canadensis* (Threatened)**
- **Woodland Caribou, *Rangifer tarandus caribou* (Endangered)**
- **Bruneau Hot Springsnail, *Pyrgulopsis bruneauensis* (Endangered)**
- **Bliss Rapids Snail, Taylorconcha serpenticola (Threatened)**
- **Utah Valvata, *Valvata utahensis* (Endangered)**
- **Banbury Springs Lanx, *Lanx sp.* (Endangered)**
- **Snake River Physa, *Physa natricina* (Endangered)**
- **Goose Creek Milkvetch, *Astragalus anserinus* (Candidate)**
- **Christ's Indian Paintbrush, *Castilleja christii* (Candidate)**
- **Water howellia, *Howellia aquatilis* (Threatened)**
- **Slick Spot Peppergrass, *Lepidium papilliferum* (Threatened)**
- **MacFarlane's Four O'clock, *Mirabilis macfarlanei* (Threatened)**
- **Spalding's Silene (Spalding's Catchfly), *Silene spaldingii* (Threatened)**
- **Ute Ladies' Tresses, *Spiranthes diluvialis* (Threatened)**

As our human population grows and demands on natural resources increase, our society will confront unpleasant choices. Do we really value clean water, clean air, open space and abundant wildlife enough to choose smaller personal vehicles, higher energy costs, higher food costs, more restrictions on personal property use or development (Figure 1). These are societal decisions that we and our successors will participate in, by choice or by default.

Figure 1. New York Times article - Waning public support for environmental issues.

Further, social values for wildlife and wild landscapes are changing as our society moves farther from its agrarian roots. The social “contract” between the American public and wildlife managers a generation ago – to ensure adequate supplies of wild game, fish and habitat to sustain hunting, fishing, and trapping - is frequently questioned by some of today’s public who have very different wildlife values or who value other resource uses more than traditional wildlife and habitat conservation objectives.

Our world is becoming more crowded with people and expectations. The need for resource conservation is growing regardless of conflicting values. Resources will be necessary to satisfy the needs and values of humans. How those needs are satisfied, will challenge natural resource stewards (especially wildlife managers and conservationists) and beneficiaries.

Diverse Social Values Challenge Resource Managers

Because our wildlife is a common trust resource, wildlife managers in each state are responsible to the full citizenry of their respective states. Within one generation of contemporary wildlife managers, that citizenry that broadly identified with and supported “traditional” hunting and fishing management objectives and programs is no longer as unified in support of traditional management philosophies. The polar extremes of American public philosophies towards the values and acceptable uses of our wildlife resources (Figures 2, 3) have been represented in public dialog since the beginnings of the American conservation movement. However, the continued urbanization of American society, away from our country’s rural agrarian beginnings, is reshaping public values and attitudes towards nature and the role of human society in the “natural world”.

Figure 1. Pittsburgh Tribune-Review article - PETA objections to Groundhog Day tradition.

Figure 3. The Washington Times article - Fish are valued more than people.

Aldo Leopold famously commented on the distancing of "modern" American society from the land (nature) in "A Sand County Almanac":

"There are two spiritual dangers in not owning a farm. One is the danger of supposing that breakfast comes from the grocery, and the other that heat comes from the furnace.

….our educational and economic system is headed away from, rather than toward, an intense consciousness of land. Your true modern is separated from the land by many middlemen, and by innumerable physical gadgets. He has no vital relation to it; to him it is the space between cities on which crops grow." (Leopold 1949)

Those observations were inspired by American society immediately after WWII, over 60 years ago. That evolution, and distancing, has continued unabated and presents wildlife managers and conservationists with concomitant evolving public values and expectations to reconcile with traditional wildlife management objectives and programs to achieve those objectives.

As our population and its demands for natural resources inexorably increases, the habitat conservation requirements and restrictions on human activities (some which will be considered by many as fundamental freedoms) necessary to conserve native species and other wildlife will force hard choices. For wildlife conservationists, the evolving social landscape will require compelling arguments for the value of abundant wildlife and habitat if those choices are to be on the side of native species and the North American wildlife heritage.

Whither Native Species Conservation?

What are the most effective strategies for native species conservation? Are there strategies or messages that should be avoided? Some argue that in our passion for the conservation of native species, wildlife and other natural resources, we as wildlife management professionals may misuse our scientific credibility to the detriment of wildlife conservation goals.

"Unfortunately, science is increasingly misused in policy analysis and decision making, even by scientists;….I argue that unless we are more vigilant guarding against the misuse of science in natural resource policy and management, we risk marginalizing the helpful role that science and scientists can play in resolving important, but divisive natural resource issues." (Lackey 2009)

As the choices between the necessary sacrifices to conserve abundant wildlife and habitat, and cheap energy, affordable clean water, abundant jobs become starker, the credibility of conservation leaders with the public will be crucial to successfully argue for meaningful conservation of native species. Erroneous and counter-productive messages, especially by natural resource public servants and private sector conservationists, must be avoided if we are to persuade fellow citizens and elected leaders that native species conservation should be a local, regional and national priority.

Statements such as

- "I work for the resource."
- "It's time that society concedes the needs of native species. They were here first."
- "Putting people first is species-centric."
- "The human species is the biggest threat to life on earth."

are in fact counter-productive to native species and broad wildlife conservation goals. No wildlife resource professional has ever worked a minute "for the resource". Wildlife resources, including native species, are just that – resources managed for a diversity of benefits - for society. The human species may be a threat to itself, but "mother earth" will continue to prosper regardless of our own future.

"We can surely destroy ourselves, and take many other species with us, but we can barely dent bacterial diversity and will surely not remove many million species of insects and mites. On geologic scales, our planet will take good care of itself and let time clear the impact of any human malfeasance." (Gould 1993)

We who are privileged to steward public wildlife resources must understand we work for the public and that our responsibilities are founded on the needs and desires of society – the Public Interest.

There are No Simple Solutions

Ultimately, it is society that "chooses" native species priorities.

"Given the predicted human population increase, the over-all, long-term, downward trend in wild salmon abundance is nearly certain unless there are spectacular changes in the life styles of the region's inhabitants; … But, apart from equivocal polling data, opaque political rhetoric, and grand statements of intent, there is little tangible evidence that most people are willing to make the substantial personal or societal changes needed to restore large runs of wild salmon; … I contend that the future of wild salmon is not hopeless or foreordained, but society has collectively shown scant willingness to adopt the policy choices necessary to reverse the long-term downward trend in wild salmon." (Lackey 2001)

Effective native species conservation strategies must be convincing and compelling, and in the interests of individuals if society is to make native species a necessary priority. Personal interactions with native species - through fishing, wildlife watching or hunting - are assets that strengthen the message of value of native species for society. For many Americans, simply knowing that wild lands and wildlife exists is a reassuring priority with significant value. For many other Americans, there is a perspective that somehow conservation of wildlife and wildlife habitat is a choice between human values and the intrinsic "needs" of animals, individually or collectively. The simplistic notion that conserving water for the conservation of salmon, and therefore restricting the use of water for municipal or agricultural use, is "anti-human" is a common refrain (Figure 3).

We must do more to clarify that conservation laws are for the benefit of people and society. As the distance between the American population and nature increases, the relevance of native species and the natural world, of which we are a part, becomes easier to forget. More of our resources and efforts will be needed to help the American public understand that conservation laws and programs only exist to benefit human society.

"We have a legitimately parochial interest in our own lives, the happiness and prosperity of our children, the suffering of our fellows. The planet will recover from a nuclear holocaust, but we will be killed and maimed by the billions, and our culture will perish. The earth will prosper if polar icecaps melt under a global greenhouse, but most of our major cities, built at sea level as ports and harbors, will founder and changing agricultural patterns will uproot our populations." (Gould 1993)

Finding Common Ground

Building public support for native species conservation or any wildlife conservation initiative will be essential to the success and effectiveness of future programs and initiatives. How can the conservation community help a diverse and perhaps disaffected public conclude that native species conservation and other relatively esoteric or potentially threatening wildlife management objectives are in fact in their best interest? What will convince a northern pike fisherman in westslope cutthroat trout and bull trout country or urbanites who have never been west of the Mississippi and have never fished, hunted or even camped that they should care about the future of a native species or biological diversity? That is one of the most important challenges for the conservation community now and will continue to be for the foreseeable future.

There are contemporary examples.

Both Oregon and Idaho have pursued options under Section 4d of the ESA to continue sport fishing for bull trout populations while under ESA Threatened status, with U.S. Fish and Wildlife Service (USFWS) approved limitations. If sport angling can be allowed while achieving the protection required under ESA Threatened status, a strong incentive for support from the sport fishing community, for the conservation of bull trout, may be created.

If wildlife recreation creates economic activity that is important to local and regional economic vitality, the business community and elected leaders have a compelling incentive to support meaningful conservation goals and programs. In Idaho alone, the 2006 USFWS National Survey of Fishing, Hunting, and Wildlife-Associated Recreation found that fishing, hunting and other wildlife based recreation generated over US$1 billion in economic activity for the state of Idaho. Similar surveys conducted by the Idaho Department of Fish and Game and others confirm that fishing, hunting, and wildlife watching generate millions of dollars of economic activity for local and regional economies. The economic value of wildlife and wildlife-based recreation is a powerful incentive to support wildlife conservation, including native species.

The Idaho Department of Fish and Game and other state agencies and institutions around the country are aggressively promoting an initiative based on the bestselling book "Last Child in the Woods" (Louv 2008) to revive youth participation in outdoor activities. These programs strive to promote a re-connection with nature for our youth, based on research documenting mental, emotional, and physical health benefits derived from personal connection to nature.

These are only a few examples of strategies to demonstrate to the average American that wildlife, including native species, are a valuable national asset worthy of conservation and as priorities to choose when difficult choices become unavoidable. A more comprehensive list of benefits and values of native species will be invaluable for the coming public campaigns to conserve American wildlife.

Public Policy – Management Reality

Abraham Lincoln:

"No policy that does not rest on some philosophical public opinion can be permanently maintained"

Speech at New Haven

"Public sentiment is everything. With public sentiment, nothing can fail. Without it, nothing can succeed. Consequently, he who molds public sentiment goes deeper than he who enacts statutes or pronounces decisions."

First Debate with Douglas

Our 16th President was more than an exceptional national leader. He was a consummate politician with a masterful understanding of the public and the fundamental necessity of public support to successfully govern. The same principles that he described as essential to achieve a national policy to abolish slavery applies equally to efforts to conserve wildlife, wildlife habitat and American traditions of fishing, hunting, and other wildlife-based recreation. Without the agreement with and commitment to that fundamental premise from our society, there will be no foundation for a choice in favor of native species and abundant wildlife. If the American public perceives native species or other wildlife conservation goals to require serious sacrifice of jobs, affordable energy or food, WITHOUT personal benefit – the public decision will be easy. It will not, however, favor native species conservation. If the American public concludes that conserving wildlife and habitat result in a better life for this and future generations, then support for the tough choices that favor abundant wildlife will be much more likely.

Literature Cited

Czech, B., and P. R. Krausman. Research Notes: Public Opinion and Endangered Species Conservation. Society & Natural Resources 12(5): 469 – 479.

Economy, Jobs Trump All Other Policy Priorities In 2009. Pew Research Center. January 22, 2009.

Gould, S. J. 1993. The Golden Rule: A Proper Scale for Our Environmental Crisis. Page 49 *in* Eight Little Piggies. W.W. Norton, New York.

Lackey, R. T. 2001a. Defending Reality. Fisheries 26(6): 26-27.

Lackey, R. T. 2001b. Salmon and the Endangered Species Act: troublesome questions. Renewable Resources Journal 19(2): 6-9.

Lackey, R. T. 2001c. Values, policy, and ecosystem health. Bioscience 51(6): 437-443.

Lackey, R. T. 2009. Interpreting Naturalness in Science and Policy. Northwest Science 83(3): 277-279.

Leopold, A. 1949. A Sand County Almanac. Oxford University Press, London, Oxford, New York.

Locke., E. A. Animal "Rights" Versus Human Rights. Intellectual Conservative. June 2, 2005.

Louv, R. 2008. Last Child in the Woods. Algonquin Books of Chapel Hill, Chappel Hill, North Carolina.

Middleton, Brandon. Reality bites: the Endangered Species Act puts fish before people. PLF Liberty Blog. July 13, 2009.

Obegi, D.. The Fallacy of "Fish versus People". Living Sustainably. July 8, 2009.

Revkin, A. C. Environmental Issues Slide in Poll of Public's Concerns. The New York Times. January 23, 2009.

Richardson, V.. It's farmers vs. fish for California water. The Washington Times. August 20, 2009.

Togneri, C. PETA wants animatronic groundhog: Who are they kidding? Pittsburgh Tribune-Review. January 28, 2010.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation.

Becoming Trash Fish: The 20th-Century Marginalization of Mountain Whitefish

Jennifer Corrinne Brown
Ph.D. Candidate, Department of History, Washington State University

ABSTRACT — This paper uses mountain whitefish *Prosopium williamsoni* to investigate the history of western sport fishing and conservation. Throughout the 20th century, anglers, regional boosters, and the hatchery system created a trout aesthetic, refined on eastern rivers from its European roots and superimposed on the western landscape. In some western trout waters, mountain whitefish serve an important role as an indicator species. Mountain whitefish can also act as an indicator of the current difficulties in protecting native fish in western rivers that have become the foundation for a lucrative sport fishing industry that relies on nonnative trout. This paper will investigate the creation of a western trout aesthetic that has celebrated nonnative fish, to the detriment of mountain whitefish and other native species. Historical research from angling literature, scientific studies, government documents, and regional booster literature will shed light on contemporary problems surrounding mountain whitefish and native fish conservation in general.

Introduction

Former *Rocky Mountain News* outdoor reporter Ed Dentry once recalled a disturbing sight on the Madison River. He watched a fisherman become more and more irritated by catching numerous mountain whitefish *Prosopium williamsoni*. The fisherman then landed a large whitefish and threw it on the bank. Dentry described the next scene: “He started his next cast, then changed his mind. He rushed up the bank, found the flopping fish and jumped up and down on it” (Dentry 2007). The reporter saw this as “typical” of the species’ poor reputation among anglers (Dentry 2007). Throughout the 20th century, many sport fishers have despised catching this native western fish, but have done so frequently because of the fishes’ abundance in the region’s world-famous trout streams.

In some western trout waters, mountain whitefish serve an important role as an indicator species, alerting biologists to ecological problems like pollution or low water temperatures that may endanger other fish populations or drinking water sources (Varley and Schullery 1983; Meyer et al. 2009). Mountain whitefish can also act as an indicator of the current difficulties in protecting native fish in western rivers that have been transformed into lucrative sport fishing destinations that rely on nonnative trout. This paper will investigate the creation of a western trout aesthetic that has celebrated nonnative fish, to the detriment of mountain whitefish and other native species.

The inattention to mountain whitefish represents a historical trend among anglers and fisheries managers. Fish authorities have neglected mountain whitefish in three distinct, yet overlapping and competing, time periods in fish management: (1) the hatchery era, beginning in the 1850s and becoming institutionalized until at least the 1970s; (2) the “wild trout” era, starting in the 1960s and 1970s, which focused on nonnative game fish that had become the mainstay of the western fishing industry; and, (3) the native fish era that currently centers on native species and biodiversity. During the first two eras, anglers, their license fees, and their fishing culture shaped fisheries management (Schullery 2006).

The Hatchery Era

During the hatchery era, federal, state, and private hatcheries sought to save declining fish populations and improve sport fisheries by stocking billions of fish and introducing nonnative trout. Western waters became recreational spaces as conservationists enacted regulations and stocked preferred trout species, largely based upon Euroamerican sporting traditions. A prejudice against coarse fish, simply defined as other species besides salmon and trout, existed within parts of English and American fishing culture and influenced western hatchery decisions. Francis (1863) observed a British dislike for coarse fish. This certainly shaped his work as the piscicultural director for the English Acclimatization Society, an organization that introduced new species of plants and animals throughout the nation (Norris 1868). As an elite angler, Francis (1863) connected coarse fish to poorer classes and Jews in England . Similarly, the author of one of the earliest American fishing books, Scott (1869) portrayed fishing as a more democratic venture in America but still privileged eastern brook trout over other fish: "Trouting is an abiding and universal source of pleasure to all classes and conditions of men and boys—ay, and of ladies also." Scott (1869) called coarse fish "leather-mouthed fishes" and stated they "are not generally regarded as gamy, though good sport for ladies and youth". Although he did see a place for coarse fish within the sport, Scott based his view on the assumption that they could only be enjoyed by seemingly inferior anglers (See Figure 1). The disdain for coarse fish (later called "trash fish") became an element of western fishing culture as the nation expanded westward. Mountain whitefish and some other native western fish shared similar appearances and thus comparable prestige with the coarse fish of England and the eastern United States. These ideas influenced the hatchery system, which sought to remake western waters by introducing favored brook trout *Salvelinus fontinalis*, rainbow trout *Oncorhynchus mykiss*, and brown trout *Salmo trutta*.

Scientists in the fledgling U.S. Fish Commission observed angler likes and dislikes and always promoted their work spreading the ever-so-popular trout species. A reverend turned fish culturist, Livingston Stone explained in 1873 why brook trout were the most popular freshwater game fish of the time: "He surpasses all other fish in grace of form, in beauty of coloring, in gentleness of expression, in fascination of manner, in gameness of spirit, in sweetness and firmness of flesh, and in general personal attractiveness…" (Stone 1873). Thirty years later, scientists David Starr Jordan and Barton Warren Evermann (1902) discussed how the commission distributed popular game fish throughout North America and the world. They noted that anglers now considered

Figure 1.—"Girl Fishing for Roach" (Source: Norris 1864). Early prejudices against coarse fish (such as roach) influenced western fishing culture and fisheries management, as some western species like mountain whitefish bore resemblance to the coarse fish maligned in Europe and the eastern United States.

rainbow trout "the great of all game fishes" because of their "beauty of color, gracefulness of form and movement, [and] sprightliness when in the water…" Considerations of anglers' preferences for certain species guided the rearrangement of western sport fisheries during the hatchery era.

How did mountain whitefish fare during the hatchery era? The first superintendent of the Bozeman federal fish hatchery, Dr. James A. Henshall (1902), revealed the species was "not so highly esteemed" among western anglers. While resource managers introduced and artificially propagated almost every conceivable species in waters across the nation, they rarely produced mountain whitefish that were disliked by sport fishers. Instead, fisheries management centered on creating more trout and other esteemed game fish. Henshall (1919) later lamented the decline of native fish in the upper Missouri River basin and questioned his role in introducing nonnative fish, stating that "there was no good reason or valid excuse, except that applicants asked for brook, rainbow or steelhead trout, and they were supplied." Yet few of Henshall's contemporaries, or even later managers and anglers, decried the introductions of nonnative fish. Rather, they celebrated the ability to improve nature and promoted the West as a trout paradise.

Regional Boosterism and the Creation of a Trout Aesthetic

Environmental and western historians have sometimes used bioregionalism to define regions by their unique flora and fauna. Journalist Timothy Egan has been widely quoted as defining the Pacific Northwest as "wherever the salmon can get to." To define a region based on "wherever the trout can get to" becomes a bit problematic. With the exception of Antarctica, rainbow trout and brown trout have been introduced to every continent for sport fishing purposes (Schullery 2006: 198-201; Crawford and Muir 2008). Few places in the world, however, have advertised their trout fishing opportunities like the American West. Railroad companies, chambers of commerce, state agencies, and the fishing industry have all sold the region as a premier trout destination.

Railroads transported trout to the region and hoped to profit from the tourism it produced by publishing various pamphlets and brochures about western fishing. The best fishing depended, of course, on which rail line that tourists traveled. The Colorado and Southern Railway (1907) championed the Colorado trout streams that ran along its lines. The Great Northern (1901) believed the "finest of trout fishing is to be found in western Montana, Idaho and Washington." And the Union Pacific (1908) argued that Wyoming had "the finest trout fishing to be found anywhere on the globe…" As railroad companies helped federal and state hatcheries stock fish, their circulars started a publicity campaign that connected the West with nonnative trout.

Local and state governments also promoted western trout fishing. A key selling point in a 1940 Montana brochure proved to be the abundant fishing: "Montana has 32,000 miles of trout streams, 4,600 miles of improved highways" (Her Majesty Montana 1940). In his book *Western Trout*, fishing author Syl MacDowell (1948: 174) called trout "big business" in the West. He noticed that by mid-century western trout had become an economic and political issue. By mid-century, trout species thrived in western coldwater rivers and supported a sizable tourism and fishing industry. Anglers, hatcheries, and regional boosters created a trout fishing aesthetic, refined on eastern rivers from its European roots and superimposed on the western landscape. As anglers exalted trout in their new western habitats, some disdained mountain whitefish and other native species.

Becoming Trash Fish

The evolution of a western trout aesthetic translated into poor treatment of mountain whitefish. The species was perhaps the most reviled native western fish and anglers considered them "trash fish." In the 1930s, California Fish and Game officials observed Truckee River anglers disliked mountain whitefish: "At well-fished 'holes' it is not uncommon to see many whitefish strewn on the banks to rot. Many anglers throw away their entire catch of this species—believing it to be worthless" (Dill and Shapovalov 1939). The authors noted anglers often mistook the species for suckers (*Catostomidae* family), even though mountain whitefish are more closely related to trout. The article provided an illustrated guide to help anglers distinguish between species but offered little information about who would throw away these fish. As anglers sought western trout, they increasingly came into contact with less-prized native species like mountain whitefish.

Similar episodes occurred throughout the West during the mid-century. The Wyoming state fish warden reported in 1941 that during the summer, anglers left piles of rotting mountain whitefish along the banks of the Salt, Snake, Shoshone, and Upper Wind rivers (Simon 1941). A 1946 British Columbia guidebook informed readers that anglers frequently caught the species, but threw them away because they resembled "coarse fish" (Pochin 1946). Harriet Wheatley, a well-to-do tourist who wrote about her extensive travels in 1952, recounted that her and her husband rarely kept the species. At the behest of a guide in Jackson Hole, they supplied mountain whitefish to less discerning tourists staying in a local lodge. Wheatley and the other members of their fishing party delighted in filling their creels and she reported that, surprisingly, the mountain whitefish fought well and tasted good (Wheatley 1952). These authors noted mountain whitefish had similar qualities compared to trout in terms of game, food, and fishing opportunities. If so, why were they so stigmatized among anglers?

Mountain whitefish are unfortunate-looking creatures. The perceptions of the fish have also been shaped by anglers' sensory experiences and reactions. Mighetto (1991) notes that, to some extent, anthropomorphism still influences animal protection and many conservation groups concentrate on animals who exhibit human-like features. While mountain whitefish may not be on the hideously-ugly level of deep sea creatures or microscopic views of insects, but they are less attractive than other trout species prized for their beauty and game qualities. Their small, protruded mouths with no teeth, and the lack of pleasing colors and spots, differentiates mountain whitefish from the more well-liked trout species. Many anglers have discussed the unbecoming qualities of mountain whitefish, particularly its snout and scaly, brownish-grey body. Even Greg Keeler sings the "White Fish Blues," in which he compares the snout to something "you'd use to vacuum out your car." The unbecoming appearance of mountain whitefish created significant consequences for this particular species, as some anglers and fisheries managers have ignored or under-appreciated the fish.

As western waters became linked to a trout aesthetic, anglers expected a certain fishing experience. The presence of mountain whitefish tended to annoy some trout seekers. "In many waters I have found them to be so plentiful and pestiferous that they spoiled fishing," Syl MacDowell complained in 1948, "My principal grievance against all whitefish is that they arrange to be caught when I have an expectant and critical audience whom I prefer to impress by connecting with a big trout" (MacDowell 1948). Fish managers also charged the fish with competing with trout for valuable food and space. Although some food overlap occurred, fisheries biologists assumed that removing mountain whitefish would increase trout populations in rivers (for example, see Sigler 1951). Meyer et al. (2009) have observed that even though some agencies engaged in mountain whitefish removal between the 1950s and 1970s, the idea that the species compete with trout "has not been substantiated." Fish managers did not need proof that fewer mountain whitefish would increase trout populations, all they needed to know was that anglers desired to fish for trout.

The Wild Trout Era

In the second era of western sport fisheries management, anglers, some states, and certain agencies began to question the consequences and costs of decades of fish culture and hatcheries. By this time, nonnative trout had entrenched themselves in western waters and managers sought to protect the nonnative species who became an important part of the West's tourism and recreation industries. Some states ended river stocking and scaled down their hatchery systems while still concentrating on nonnative species. Supported by new conservation groups like Trout Unlimited and the Federation of Fly Fishers, state agencies lowered daily catch limits to promote wild trout management. How did the unappreciated mountain whitefish make out during the wild trout era? Certainly new concerns with habitat and conservation benefited the species, but as fisheries managers continually restricted the catch of trout, they raised the daily catch limits for mountain whitefish (DosSantos 1985). Or, they continued with mountain whitefish removal programs (Meyer et al. 2009). Yet despite the years of habitat change in the industrial and agricultural West, the mistreatment by many anglers who considered them trash fish, and the lack of management, with some key exceptions, they have survived.

The Native Fish Era

Many western waters like the Madison still contain sizable populations. However, they are on the verge of disappearing from the Big Lost River in Idaho. By 2005, the adult mountain whitefish numbers in the river had dropped to only 1.5% of their historic population (Idaho Department of Fish and Game 2007). As the focus on wild trout and the development of conservation biology led to a great emphasis on native fish and preserving biodiversity, fish managers are now forced to confront the past. Has this growing interest for native fish included the sullied mountain whitefish? Based on a firmly planted western trout aesthetic, the species remains ignored by managers and anglers. Indeed, Meyer et al. (2009) note that the species "continue to remain an afterthought for most fisheries research and management programs in western North America." Anglers' mistreatment of mountain whitefish has also continued to the present, as shown by Ed Dentry's whitefish stomping story. Although discontinued in the early 2000s, the annual whitefish festival held on Colorado's Roaring Fork River demonstrated the fish's poor reputation. As if in some type of bizzaro-world, anglers took advantage of the state's lack of regulations and always pulled hundreds of whitefish out of the river. Corporate sponsors then replaced the dead fish with hatchery rainbow trout (Dentry 1999). The 20th-century development of a western trout aesthetic translated into a certain amount of mistreatment or neglect of native fish species. Despite having unprivileged status as an ugly creature, the conservation of mountain whitefish and other so-called trash fish continues to be crucial for western rivers.

References

Colorado and Southern Railway. 1907. Trouting in Colorado's waters. Smith-Brooks, Denver.

Crawford, S. S. and A. M. Muir. 2008. Global introductions of salmon and trout in the genus *Oncorhynchus*: 1870-2007. Reviews in Fish Biology and Fisheries 18: 313-344.

Dentry, E. 1999. Not-so-great whitefish still offers a little hope. Rocky Mountain News (October 6).

Dentry, E. 2007. Lowly whitefish deserves respect – and research. Rocky Mountain News (February 7).

Dill, W. A. and L. Shapovalov. 1939. An unappreciated California game fish, the Rocky Mountain whitefish, *Prosopium williamsoni* (Girard). California Fish and Game 25: 226-227.

DosSantos, J. M. 1985. Comparative food habits and habitat selection of mountain whitefish and rainbow trout in the Kootenai River, Montana. Master's thesis. Montana State University, Bozeman.

Francis, F. 1863. Fish-culture: A practical guide to the modern system of breeding and rearing fish. Routledge, Warne, and Routledge, London.

Great Northern Railway. 1907. Shooting and fishing along the line of the Great Northern Railway. 4th ed. Poole Bros., Chicago.

Henshall, J. A. 1902. Food and game fishes of the Rocky Mountain region. Transactions of the American Fisheries Society 31: 74-88.

Henshall, J. A. 1919. Indiscriminate and inconsiderate planting of fish. Transactions of the American Fisheries Society 48: 166-169.

Her majesty Montana has a "come-hither" look. 1940. Montana Standard Printers, Butte.

Idaho Department of Fish and Game. May 2007. Mountain whitefish conservation and management plan for the Big Lost River drainage, Idaho. Available at: http://fishandgame.idaho.gov/fish/programs.whitefish.pdf. (September 2008).

Jordan, D. S. and B. W. Evermann. 1902. American food and game fishes: A popular account of all the species found in America north of the equator, with keys for ready identification, life histories and methods of capture. Doubleday, Page, New York.

MacDowell, S. 1948. Western trout. Alfred A. Knopf, New York.

Meyer, K. A., F. S. Elle, and J. A. Lamansky, Jr. 2009. Environmental factors related to the distribution, abundance, and life history characteristics of mountain whitefish In Idaho. North American Journal of Fisheries Management 29: 753-767.

Mighetto, L. 1991. Wild animals and American environmental ethics. University of Arizona Press, Tucson.

Norris, T. 1864. The American angler's book: Embracing the natural history of sporting fish, and the art of taking them. with instructions in fly-fishing, fly-making, and rod-making; and directions for fish-breeding. E. H. Butler, Philadelphia.

Norris, T. 1868. American fish-culture, embracing all the details of artificial breeding and rearing of trout: The culture of salmon, shad and other fishes. Porter & Coates, Philadelphia.

Pochin, W. F. 1946. Angling and hunting in British Columbia. Sun Directories, Vancouver.

Schullery, P. 2006. Cowboy trout: western fly fishing as if it matters. Montana Historical Society Press, Helena.

Scott, G. C. 1869. Fishing in American waters. Harper & Brothers, New York.

Sigler, W. F. 1951. The life history and management of *Prosopium williamsoni* in the Logan River, Utah. Utah State Agricultural College Bulletin 347. Utah State Agricultural College, Logan.

Simon, J. R. 1941. The whitefish—A neglected resource. Wyoming Wild Life 6: 1-3.

Stone, L. 1873. Domesticated trout: how to breed and grow them. James R. Osgood, Boston.

Union Pacific Railroad. 1908. Union Pacific outings: Fishing in Colorado and Wyoming. n.p.

Varley, J. D. and P. Schullery. 1983. Freshwater wilderness: Yellowstone fishes and their world. Yellowstone Library and Museum Association, Yellowstone National Park.

Wheatley, H. 1952. Lady angler: Fishing, hunting, and camping in wilderness areas of North America. Naylor, San Antonio.

TROUT STOCKING: A CATALYST FOR THE ENDANGERED SPECIES ACT

M. Anders Halverson
Research Associate, University of Colorado Center of the American West,
halverson@andershalverson.com

ABSTRACT — It is a well-publicized fact that the Endangered Species Act drives many fisheries management decisions today, especially those pertaining to hatcheries and stocking. Less well known is the role that recreational fisheries management played in the creation of that act. In 1962, state and federal agencies poisoned the fish in the Green River in Wyoming and Utah to further their goal of turning the almost finished Flaming Gorge reservoir into a first-rate rainbow trout *Oncorhynchus mykiss* fishery. The controversy that ensued prompted Secretary of Interior Stewart Udall to declare that in the future, the threat such operations posed to a species as a whole would have to be a dominant consideration when evaluating whether to proceed. Several months later, Udall created a new body, the Committee on Rare and Endangered Wildlife Species, and charged it with protecting endangered wildlife. Here, I describe these events, the ways in which they ultimately led to the Endangered Species Act, and transformed fisheries management today.

INTRODUCTION

The Endangered Species Act of 1973 (ESA) is one of the most important single laws affecting fisheries management today. Many hatchery and stocking programs have been curtailed or reformulated to comply with the act or forestall the listing of a candidate species (e.g. LaBar and Frew 2004; Rahel 2004). Numerous wild trout fisheries are a direct or indirect result of this law and rare is the manager, angler, or interested citizen who is not familiar with it. Less well known, however, is the important role that recreational fisheries management played in the creation of the act. Here I will summarize the efforts by state and federal agencies to eliminate wild fish in the Green River basin in 1962 and restock it with rainbow trout *Oncorhynchus mykiss* and other game fish. I will describe the national controversy that ensued, and the role those events played in the passage of the ESA. Anyone wishing to know more about these events or see a broader list of sources should refer to my book, *An Entirely Synthetic Fish* (Halverson 2010), or to Holden (1991) or Wiley (2008).

THE GREEN RIVER "REHABILITATION"

On September 4, 1962, officials with the Wyoming Department of Game and Fish, the Utah Department of Fish and Game, and the U.S. Fish and Wildlife Service began what they referred to as the "rehabilitation" of the Green River above Flaming Gorge in Wyoming and Utah. For three days and nights, they used drip lines, airboats, and a helicopter to pour more than 80,000 liters of rotenone into the river and its tributaries. Their goal was to kill any fish that might interfere with their goal of creating a world-class trout fishery in the watershed (Binns et al. 1963).

Affecting 724 km of river draining a 38,000 km^2 watershed, the Green River operation was the largest of its kind up to that time, but it was certainly not the first. In the decade previously, state and federal agencies had used rotenone to eliminate wild fish from 4,000 km of running water and 920 km^2 of lakes in 34 different states. California started things off by poisoning the Russian River in 1952. Montana followed suit in the Marias and the Clark Fork rivers. In the South, Abrams Creek in Great Smoky Mountains National Park was poisoned to make way for the introduction of rainbow trout (Lennon et al. 1971; Miller 1963).

Fisheries managers in Wyoming and Utah conceived of the Green River operation when they learned that the Bureau of Reclamation planned to build a large dam on the Green River in Utah's Flaming Gorge and another smaller one 240 km upstream in Wyoming. The managers knew the river would undergo a fundamental change once the dams were completed—among other things it would be colder, would have more even year-round flows with less sediment, and there would be large bodies of standing water behind the dams. They saw it as an opportunity to create a terrific trout fishery. But they also feared that the carp *Cyprinus carpio* already present in the river would find ideal habitat in the reservoirs and their population would explode (Binns et al. 1963).

From the point of view of its proponents, the operation was a tremendous success. More than 400 metric tons of fish were killed and later surveys showed that virtually none were left alive in the river. The aquatic invertebrates were also nearly totally destroyed. Over the succeeding years, the agencies stocked rainbow trout, kokanee salmon *Oncorhynchus nerka*, and other game fish into the river and its reservoirs, and today the area is known as one of the best trout fisheries in the country. There were some unintended consequences to the operation, however. (Binns et al. 1963)

The Bureau of Reclamation had initially told the fish and game agencies that it was going to shut the gates on the Flaming Gorge Dam in September, and the agencies had planned to begin the rotenone operation shortly thereafter. They reasoned that the rotenone would be trapped behind the dam, where natural processes would detoxify it before it could flow any further downstream. When the Bureau of Reclamation decided to delay the closing of the gates until November, the agencies were faced with a quandary. At that time of year, the water would be too cold for the rotenone to be effective. They decided instead to proceed with the operation as planned, and to use potassium permanganate ($KMnO_4$) to detoxify the rotenone. They found a bridge in Browns Park, Colorado, just over the border from Utah, from which to pour the $KMnO_4$ and so were able to begin the operation in September as originally planned. (Binns et al. 1963)

The detoxification effort, however, ran into difficulties from the beginning. The $KMnO_4$ was the wrong consistency, making it difficult to apply; the rotenone concentration was higher than expected when it reached the bridge, which meant they did not have enough $KMnO_4$; and a cold front arrived, bringing severe weather that made working conditions less than ideal. As a result, rotenone slipped by the bridge, and a few days later dead fish were found downstream in Dinosaur National Monument (Binns et al. 1963).

The Controversy

Hard to believe as it may be in this day and age, there was virtually no controversy about the operation while it was being planned. Two ichthyologists, Robert R. Miller and Carl Hubbs, tried to stir up opposition to the project because of the effects it would have on native fish like the humpback chub *Gila cypha*, bonytail *Gila elegans*, razorback sucker *Xyrauchen texanus*, and Colorado pikeminnow *Ptychocheilus lucius*, but their efforts were largely in vain. What newspaper stories did get written were often buried in the sports pages and gave short shrift to the opponents. Even the environmental organizations were curiously silent on the matter. The plight of native species was not high on the American radar screen on the time, and especially not the plight of native fish. (e.g. Hubbs 1961; Denver Post 1962)

After the dead fish were found in Dinosaur National Monument, though, everything changed. Dinosaur National Monument had become hallowed ground to many Americans in the decade previous, after a widely publicized battle about whether the country should build a dam there. The opponents of the dam won, thanks to a public relations campaign that convinced many Americans that our National Parks and Monuments were national treasures. (Fox 1981)

Equally important, Rachel Carson published Silent Spring about three weeks after the Green River operation was completed. This epochal book provoked national concern and even outrage over the widespread use of insecticides, herbicides, and other chemicals (Carson 1962).

It remains unclear whether fish were killed in the monument, or whether the fish had been killed upstream and only washed up there. But that became a moot point. The combination of chemicals, dead animals, and a national monument created a volatile combination and Miller, Hubbs, and their allies finally began to get some traction (Binns et al. 1963).

Letters arrived on the desks of congress members, demanding answers about just what had happened. The chairman of the House Natural Resources and Power Subcommittee passed the demands on to Interior Secretary Stewart Udall, demanding to know whether he really believed his department had "fully complied with its responsibilities." Officials with the National Park Service and the U.S. Fish and Wildlife Service blasted each other over the affair, and finally Secretary of Interior Stewart Udall began a formal review (Holden 1991).

The Endangered Species Act

In the report he issued in the spring of 1963, Udall, declared that from then on "Whenever there is question of danger to a unique species, the potential loss to the pool of genes of living material is of such significance that this must be a dominant consideration in evaluating the advisability of the total project" (Udall 1963).

Shortly thereafter, Udall created a new body, the Committee on Rare and Endangered Wildlife Species and charged it with "halting the further disappearance of endangered wildlife." Daniel Janzen, the man who oversaw the Green River operation for the federal government became the head of the new committee (U.S. Department of Interior 1964).

Much of the committee's initial focus was on compiling a list of rare and endangered wildlife and the habitats they needed to survive. The committee turned to experts from all over the country to produce this list, including Miller and Hubbs (Janzen 1964).

Thanks in large part to Udall and the efforts of the committee, Congress took up the matter in 1966, passing a law called the Endangered Species Preservation Act. The law was amended in 1969 with the Endangered Species Conservation Act to broaden its scope. Within a few years, though, this, too, was deemed inadequate and in shortly thereafter, President Nixon signed a third version, the Endangered Species Act of 1973, into law (Bean and Rowland 1997).

I do not wish to claim that the controversy engendered by the Green River operation was the sole source of the Endangered Species Act of 1973. Stewart Udall did not remember the controversy when I spoke to him in 2006, and many other things happened between 1962 and 1973 to push environmental issues into the limelight. Moreover, the U.S. Fish and Wildlife Service had been concerned with the plight of threatened species for years before the events of 1962; it was the focus of the National Wildlife Refuge system.

However, in the months that followed the Green River poisoning and the establishment of the Committee on Rare and Endangered Wildlife Species, these efforts took a quantum leap forward. In the archives of the Smithsonian Institution in Washington D.C., there is a 1964 letter from John W. Aldrich, a U.S. Fish and Wildlife Service official who had been working on similar issues for some time. "There seems to be a sudden upsurge of interest in endangered species everywhere," he declared, "including our bureau." And today, under the Endangered Species Act, the United States has spent more than $100 million trying to restore many of the very fishes that were poisoned out of the Green River in 1962 (Aldrich 1964; Hartman 2000).

In conclusion, the Green River project holds many lessons for fisheries managers today. But the most important one, I believe, is that decisions regarding recreational fisheries management often have unpredictable consequences not just in the ecological realm, but in the policy realm as well. The ultimate effects of any decision may reach far beyond the arena in which it was made, and even be at cross-purposes to the original intent.

Literature Cited

Aldrich, J.W. 1964. Letter to D.L. Allen June 1, 1964. Smithsonian Institution Archives. Accession T89021, Box 7.

Bean, M.J., and M.J. Rowland. 1997. The Evolution of National Wildlife Law. 3rd edition. Praeger, Westport, Connecticut.

Binns, A., F. Eiserman, F.W. Jackson, A.F. Regenthal, and R. Stone. 1963. The Planning, Operation, and Analysis of the Green River Fish Control Project. Utah State Department of Fish and Game and Wyoming Game and Fish Department, Cheyenne, Wyoming and Salt Lake City, Utah.

Carson, R.L. 1962. Silent Spring. Houghton Mifflin, New York.

Denver Post. 1962. "Big Fish Poison Job Slated for Wyoming." September 2, 1962. Page 7b.

Fox, S. 1981. The American Conservation Movement: John Muir and His Legacy. University of Wisconsin Press, Madison.

Halverson, A. 2010. An Entirely Synthetic Fish: How Rainbow Trout Beguiled America and Overran the World. Yale University Press, New Haven.

Hartman, T. "Fish Story." Rocky Mountain News. Dec 4, 2000

Holden, P.B. 1991. Ghosts of the Green River: Impacts of Green River Poisoning on Management of Native Fishes. Pages 43-54 in W.L Minckley and J. E. Deacon, editors. Battle Against Extinction. Native Fish Management in the American West. University of Arizona Press.

Hubbs, C.L. 1961. Letter to R.R. Miller September 23, 1961. University of Michigan Museum of Zoology Archives.

Janzen, D. 1964. Memorandum to Washington Office Directorate and Branch Chiefs; and Regional Directors. Subject: Committee on Rare and Endangered Wildlife Species. January 30, 1964. U.S. Fish and Wildlife Service, Washington, D.C.

LaBar, G.W., T. Frew. 2004. Using a Fish-Stocking Database to Demonstrate Temporal Changes in Stocking Patterns. Pages 113-118 in M.J. Nickum, P.M. Mazik, J.G. Nickum, D.D. Mack, editors. Propagated Fish in Resource Management. American Fisheries Society, Symposium 44. Bethesda, Maryland.

Lennon, R.E., J.B. Hunn, R.A. Schnick, R.E. Buress. 1971. Reclamation of ponds, lakes, and streams with fish toxicants: a review. Food and Agriculture Organization Fisheries Technical Paper (100):99

Miller, R.R. 1963. Is Our Native Underwater Life Worth Saving? National Parks Magazine May: 4-9.

Queal, C. 1962. They kill fish to make fishing better. Denver Post, 7 September, 1962, Page 23.

Rahel, F.J.. 2004. Unauthorized Fish Introductions: Fisheries Management of the People, for the People, or by the People. Pages 431-443 in M.J. Nickum, P.M. Mazik, J.G. Nickum, D.D. Mack, editors. Propagated Fish in Resource Management. American Fisheries Society, Symposium 44. Bethesda, Maryland.

Udall, S.L. 1963. Review of Green River Eradication Program. Mar 25, 1963. U.S. Department of Interior, Washington.

U.S. Department of Interior. 1964. Press Release: Janzen to Develop Program to Combat Wildlife Extinction; Gottschalk Named Director of Sport Fisheries and Wildlife, September 2, 1964. U.S. Department of Interior, Washington. Smithsonian Institution Archives T89021, Box 7.

Wiley, R.R. 2008. The 1962 Rotenone Treatment of the Green River, Wyoming and Utah, Revisited: Lessons Learned. Fisheries 33(12): 211-218.

PISCATORIAL PEDAGOGY AND THE 'GEOGRAPHY OF HOPE': REFLECTIONS ON TEACHING THE LITERATURE AND CRAFT OF FLY FISHING

William F. Grover

Professor, Political Science Department, Saint Michael's College, Colchester, Vermont, 05439. 802-654-2463. wgrover@smcvt.edu

ABSTRACT — In his classic 1960 "Wilderness Letter," Wallace Stegner offers a spirited defense of "the wilderness idea." He concludes by suggesting that wilderness "can be a means of reassuring ourselves of our sanity as creatures, a part of the geography of hope." For the past six fall semesters I have been fortunate to be able to teach in the First Year Seminar program at Saint Michael's College in northern Vermont. In so doing I have made my modest contribution to nurturing such reassurance and hope. The course began tentatively, as a raw experiment, a departure from my normal classes in American politics. It has blossomed into a highly sought-after academic offering. Along the way students have learned how to fly cast on the lawns overlooking Vermont's Green Mountains, plied the waters of several of Vermont's finest trout streams, engaged in entomological and ichthyological field work with biologists, studied the history of fly fishing, experienced the joys and difficulties of some of the greatest literature of fly fishing, learned how to tie flies, and written sometimes impassioned essays about it all. This paper will share the narrative of my experiences with these students and attempt to draw conclusions about the value of teaching the literature and craft of fly fishing as a way to encourage interest in wilderness, fly fishing and thoughtful, sustainable living.

I don't know what fly-fishing teaches us, but I think it's something we need to know.--John Gierach

INTRODUCTION

John Gierach's observation has long impressed me as an invitation to ponder why we fly fish and what, if any, deeper purposes might lurk beneath the surface of such an activity. Surely the vast and rich literature of fly fishing recognizes a dual nature to the pursuit of fish. On the one hand, fly fishing is fraught with absurdities and assorted weirdness, as Gierach's many books amply attest. Hughes (1999) characterizes it as "a ridiculous human passion" which "largely consists of not catching fish." Still, something deep is there, something that McGuane (1999) detects in the emphatic "long silences—the unproductive periods." He captures it well in this passage from "A New River":

> *Things that pass us, go somewhere else, and don't come back seem to communicate directly with the soul. That the fisherman plies his craft on the surface of such a thing possibly accounts for his contemplative nature (p. 167).*

Is there a piscatorial pedagogy that unlocks anything worthy of being deemed a real contemplative insight? Is it simply an exercise steeped in nostalgia or escape, or is there direct applicability to our world? McGuane suggests an answer might be found in the terrain of fly fishing, the water on which and in which the craft is plied, and its connection to our inner life—its geography, if you will.

A River Runs Through It: The Seminar

In his classic 1960 "Wilderness Letter" Wallace Stegner offers a spirited defense of "The wilderness idea." He concludes by suggesting that wilderness "can be a means of reassuring ourselves of our sanity as creatures, a part of the geography of hope." For the past six fall semesters I have been fortunate to be able to teach in the first year seminar (FYS) program at Saint Michael's College in northern Vermont. Through this experience I have made my modest contribution to nurturing such reassurance and hope. The course began tentatively, as a raw experiment, a departure from my normal area of expertise in American politics. All incoming students are required to take a first year seminar. Faculty are allowed considerable creative leeway to design innovative offerings as long as they adhere to some basic common characteristics: seminars must be writing intensive, they must be interdisciplinary to some degree, they must devote the first week or so to exploration of a common text assigned over the summer, they must introduce students to the norms of academic integrity, they must require some library research, and they must focus on primary texts. Seminars are capped at 15 students to facilitate writing, including revision of multiple drafts, and to provide ample opportunities for class discussion. Beyond that, the program is open to the experimentation of faculty members who can work it out within their own academic departments, since FYS faculty are thereby available to teach one less departmental course.

What emerged from my reflection and research over the course of a summer was a proposal for "A River Runs Through It: The Literature and Craft of Fly Fishing." From the outset my intent was to expose students to the storied literature and passionate practice of fly fishing. I reasoned that the topic readily lends itself to the criteria above, particularly its deep literary tradition and its inherent interdisciplinarity. The number of potential disciplines involved is large, ranging from biology, entomology, ichthyology, and ecology to explorations of history, philosophy, ethics, religion, politics, class, gender, poetry, and the craft of fly tying. I envisioned a natural synergy of the formal seminar classroom and the greater "classroom" of nature. To that end I hoped the literature would come to life if matched by real, hands-on engagement, and reciprocally, the

Figure 1. Prof. Declan McCabe examines stream samples of insect life on the Browns River.

field work and fishing would be more meaningful and enduring when reinforced through knowledge of the history and literature of fishing. Thinking about such rich and varied connections—disciplining the mind to see entwined perspectives and new ways of knowing—is invaluable. Little about our world is more readily understood through the narrow lens of just a single, isolated perspective. The issue then became how best to put it all together in a way that makes sense within the rhythm of an academic semester—meeting twice a week for 100 minutes each session. Several key components have stood out over the years.

Getting their feet wet: Entomology, Ecology, and the Practice of Fly Fishing

At its root, fly fishing is about food. Figuring out what fish eat is central to the endeavor. We begin with readings focused on entomology and the life cycle of insects, paying particular attention to mayflies, caddis flies and stoneflies. Among many articles I've used, Dave Whitlock's short essays on aquatic insects for *Trout* magazine are especially clear and informative for the nonspecialist. Gierach (1990) brings a less technical but still useful and enjoyable perspective to bear on the life cycle of insects—it is, after all, just sex, death and fly fishing. David Quammen (1988) adds his personal appreciation for trout as an indicator species, a barometer of the ecological health of the environment where trout live. I have supplemented these readings with brief films, notably "The Hatch" and "Why Fly Fishing." I also have a member of the College Biology Department give a full-class talk on stream ecology and insect life, and he guides the entomology portion of our field work. Field work encompasses the next several weekends—these are required extra sessions. For the first trip we go to the Browns River, a small stream with a healthy naturally reproducing population of many types of fish, including brown trout *Salmo trutta* and brook trout *Salvelinus fontinalis*. The biologist leads half the class in the exploration of stream samples, primarily having some students kick up the substrate while downstream other students collect and identify the loosened aquatic life captured in D-frame nets. His colleague in the Biology Department, an ichthyologist, leads the other half of the class in electrofishing, taking time to examine and identify the diversity of fish species. Halfway through, we switch tasks. These hands-on experiences help illuminate some of the articles we have read, and they serve the additional practical goal of getting students accustomed to wearing waders.

We next move right in to casting lessons and learning about the basic equipment involved in fly casting—rods, reels, lines, leaders and tippets. Virtually all students are complete beginners with a fly rod. For these activities I draw on the expertise of fellow members of the Central Vermont Chapter of Trout Unlimited. Vermont weather is notoriously fickle, but September typically is a reliably beautiful month, so all of our field work is front-loaded into the first four weeks of the semester. While it might make sense to spend more time in the field studying insect and fish habitat, the climate dictates otherwise. Two full class sessions are devoted to casting lessons on the lawns overlooking the Green Mountains. I assist the casting instructor with the inevitable questions and frustrations that arise. The second Saturday of this field work sequence is then spent in a large group fishing session on the Winooski River, where the students fish for the first time. I maintain a student-instructor ratio of 5:1, so the casting teacher is joined by another TU member, and the three of us lead this excursion. Subsequent September weekends are spent with me leading groups of five students to various sections of the Winooski River—Saturday and Sunday, two groups per day. Each student gets at least three trips to the water, and the thrill of watching them catch and release a fish never gets old.

Figure 2. Prof. Doug Facey identifies characteristics of a brook trout on the Browns River.

History

A section on the history of fly fishing is a must for situating the craft within a context that accounts for shifting political, economic and cultural values. The dilemma is where to begin and what to exclude. There are many options and the choices can be brutal. I have found Robert Hughes's *A Jerk on One End* (1999) to be both brief and fairly comprehensive. He's a clear writer who moves through the long sweep of history with just enough detail for it all to make sense. I supplement his book with Dame Julian Berners's 1496 essay "A Treatyse of Fysshynge with an Angle" and selections from Izaak Walton's 1653 classic *The Compleat Angler*—two iconic works that Hughes discusses as well. In addition to the often elusive combination of brevity and solid coverage, Hughes writes in an autobiographical style that fits in well with the next section of the seminar, and in tandem with Berners (1496) and Walton he offers our first foray into the ethics of fishing and the concept of environmental stewardship.

The Search for Self

Not only is the seminar the namesake of Norman Maclean's (1976) *A River Runs Through It*, the novella also serves as the spiritual center of our endeavors. Students are powerfully drawn to the story of young Norman's search for a sense of who he is as he struggles to connect with his family and, especially, as he grapples with his inability to help his troubled and beautiful brother Paul. The themes of self, family, religion and fly fishing are interwoven with the awe and wonder of the landscape of western Montana. The novella resonates with students on many levels, and it is a treat to watch them find new meanings and insights over the years.

Maclean is so effective and affective for them because of their own life cycle; college students are at a point in their lives when developing a sense of who they are is a full time job. That Maclean leaves them with no easy out makes his work more profound, if unsettling. He remains famously "haunted by waters," while students are haunted by the questions he leaves behind. We also view Robert Redford's film version of Maclean, but not until they have written about the novella. The power of film to dominate the memory of the printed version is distressing but real. To this masterpiece I add a story by Jan Zita Grover (1999) titled "Why Fish?" Her reflections about learning how to cast a fly while she is simultaneously haunted by the reproachful voice of her late father suggest that we all bring baggage to the water. Ernest Hemingway's (1925) classic "Big Two-Hearted River" also plays on the theme of self and memory and the struggle to construct a future while haunted by the past. And it is a delight to compare and contrast his clipped prose to the soaring flourishes of Maclean. Finally, David James Duncan's (1983) *The River Why* is a class favorite, in part because Gus Orviston's coming-of-age speaks to where the students are in their own lives, and does so against the backdrop of fishing in the Pacific Northwest. As a paean to the need for love and inner balance, it is important for the students to see Gus, the master fisherman, being "unmade" as he becomes "an undone fisherman"—his search for self leading him to make true progress as a person. He offers his readers the prospect that in the quest to find our *self* we may need to *unfind* some aspects of who we are. This need to rethink the fundamentals resonates with our current situation as people on a planet facing dire environmental troubles. For surely "nature" is no mere external force, neatly separable from us. We are part of it. We cannot "save" it without contemplating how our practices, our way of life, our *selves* threaten it.

Fly Fishing as Craft

I continue to have the students learn the basics of fly tying, near the end of the semester, when the weather outside has turned toward winter and we've been entrenched in literature for a long stretch. It's a nice break and it shows them how one can stay connected to the practice of fly fishing even when actual fishing is not possible. Maclean (1976) invites us to think about craft both by directly talking about fly tying in his novella and through passages like this:

One of life's quiet excitements is to stand somewhat apart from yourself and watch yourself softly becoming the author of something beautiful, even if it is only a floating ash (p. 43).

Here he is referring to a well-delivered cast, but the floating ash imagery is poignant, and it is evocative of fly tying as the creation of floating ashes through the assemblage of feathers, hair and thread. I want my class to see that the connection between a short story, a mayfly dun floating on some riffles in a Vermont river, and a collection of feathers, dubbed hair, wire, and chenille is something beautiful, close to magical. I have a master fly tier from TU lead these two class sessions, where we tie two basic flies per meeting (e.g. woolly buggers, Mickey Finn, hare's ear Adams, Vermont caddis), and with my intermediate skills I serve as his assistant. I also include poetry in the category of craft, and throughout the seminar I sprinkle in some poems with fishing as a theme, often ones written by the late poet and longtime Saint Michael's English professor John Engels.

The "New West": Contemporary Dilemmas

It's always a challenge to choose a final book for any course. By the end of the semester students (and faculty) are tired, and the best of intentions—to expose the students to some gem as the farewell text with all the answers—often are dwarfed by our collective fatigue. My response is to assign an "easy one," easy in the sense that it is fairly brief and written in an accessible style. I've made several attempts to get this right, but I've had my best results with Robert Lee's (1997) epistolary novel *Guiding Elliott*. Lee's bumbling protagonist, fly fishing guide Donnie Phillips, faces multiple challenges as contemporary life presses up against his more traditional "backward" looking approach to life. But underneath his politically incorrect attitudes and endearing malapropisms are some important core principles. And as he reluctantly adjusts his orientation to the role of women, his emerging role as a craftsman beyond the water, and the encroachment of development on his pristine Montana, we find real wisdom in his parting counsel to "listen to rivers." Excellent class discussions have resulted from the students pondering what we might hear if we listen to rivers.

The foregoing components of the seminar are not strictly linear; all of these themes percolate throughout the readings to varying degrees. It's really a matter of what we emphasize with each one. Seeing students make connections between Lee's glimpse of fly fishing in the landscape of modern Montana, at the end of the semester, and some observation of Maclean's that we read six weeks earlier, is exciting. As a literary backdrop to their own emerging skills as fly fishers, the readings have staying power.

Constraints of the FYS Model

The single biggest constraint of my first year seminar experience will sound familiar to any teacher: mission overload. We all desire our courses to be, at least in some sense, comprehensive. This problem is more acute in an FYS setting. Unlike an introductory course I teach within my Political Science department, there is no continuing curriculum in fly fishing, no upper division extension of these basic foundations. This is a one-shot deal. The tendency, then, is to try to do too much in one semester, leaving me with the feeling that at times we've given short shrift to pretty much everything. Lots of classic literature gets left unread. Anyone with a working knowledge of fly fishing's rich literary fare would be surprised to see this or that classic gem omitted from the syllabus. But choices have to be made given time constraints. This feeling of incompleteness is compounded by the fact that the FYS model is freighted with requirements. Students read a common text in the summer and come to our first class meeting with a paper they've written about it, and that becomes the focus of the first week of class. We've had some excellent common FYS texts: Yann Martel's *Life of Pi* (2003), Khaled Hosseini's *The Kite Runner* (2003) and Kafka's (1915) seminal story "Metamorphosis" to name a few. But the common text does not have direct, obvious connections to any seminar per se, even though the life cycle of insects is about metamorphosis. Since all seminars also are writing intensive, students must write frequently, certainly every week and often for every class session. This magnifies the time pressure within the quest for completeness, since portions of almost every class are devoted to students reading their own work aloud and offering suggestions to each other. Sometimes this writing intensive requirement creates a disjuncture in their view of the class. The field work—entomology, casting, fishing—is fun; we have a great time. But I still must remain a task master since the class serves a gatekeeper role for determining which students will be deemed "writing proficient" and which will be recommended for further writing classes. Thus the fun is tempered with serious business. We also are required to address the issue of academic integrity. Add to this my own commitment to students making at least one oral presentation of our readings to enhance their public speaking skills, and the time pressures intensify.

The front-loading of outdoor activities to take advantage of good weather presents its own related challenges. The course gets quite choppy. The first week we focus on the summer reading and writing. Then we move mostly to outdoor activities for weeks. Then we return to literature and writing for a long stretch. Then it's fly tying. Then back to literature. The shifting pace is abrupt and off-putting to some. Moreover, most of the field work takes place on weekends, often very early and always cutting into their free time. They know this going into the class, of course, but early morning gatherings on Saturdays and Sundays do wear thin and there are limits to how much of their free time on the weekends can be devoted to just one class.

None of this is offered as a complaint. The strengths of this model outweigh any weaknesses. I strongly support the emphasis on the writing process within the seminar program; writing is thinking and we need more of both. The choppiness comes with the turf. Fly fishing takes place in natural settings that can change abruptly, sometimes violently. Patience and flexibility are a must. And beyond that, the first year seminar helps us forge tight bonds with these students because of the variety of shared experiences (Belec 2010; Crawford 2010), bonds that open up important opportunities to have a lasting impact.

Outcomes and Opportunities: Toward Piscatorial Hope

Certainly none of my reflections are offered as a blueprint. Pedagogical experiments like this need to be refined to see what works and what doesn't. Broadly speaking, though, there are two sets of outcomes from my involvement with fly fishing in the classroom. First are the *tangible outcomes*. The most lasting outgrowth of the seminar was the formation of the Saint Michael's Fly Fishing Club. The club took shape under the initiative of a student who enrolled in the first seminar in 2004. Now with roughly 65 members, the club features a fall and spring trip to some of the best waters in New England. It also holds fly tying sessions, knot tying clinics and occasional movie nights. Recently the club was contacted about the prospect of becoming a TU Collegiate Sub-Chapter, an offer still in the works as part of a larger TU national outreach plan (Belec 2010). Links with the TU Central Vermont Chapter have been forged, as well as loyalties to a local fly shop. Two female students from the 2005 seminar pursued an independent study project with me the following spring, producing a well-researched 40-minute film titled "The History of Women in Fly Fishing." One of those women also made earrings out of caddis fly casings, and then proceeded to make painting whimsical-looking fish a major part of her overall artistic repertoire. The seminar also has gained the attention of College alumni, who responded to an article in the alumni magazine with offers of equipment for the club and books for what is now a growing collection of fishing volumes for our library.

Beyond these tangible outcomes are harder to quantify *attitudinal outcomes* that have affected how many of these students think about fly fishing and the window it opens into environmental issues and the need for stewardship of natural resources. These changes in perspective emerge in the conversations we have after each class and after the semester ends. They continue to be manifested in the work students do when they take subsequent Political Science classes with me. In most instances you simply cannot be on the water with a fly rod in your hand for any length of time and not want to preserve and protect rivers and fish (particularly threatened cold water species) as a treasure. Achieving a balance with and respect for nature comes to seem axiomatic to these students. Channeling Dame Juliana, Hughes (1999) puts the case bluntly: "There is no defensible ethic or esthetic of angling that doesn't center on moderation" (p. 117). But the challenges, quite obviously, are daunting—indeed, unprecedented for the human species.

Climate change and the attendant pressure it puts on our political, economic, social institutions—as well as the environmental fabric of the planet—poses potentially catastrophic dangers to our ecosystem that will continue even if nations were to take immediate and drastic coordinated action to redress them (McKibben 2007 and 2010; Kolbert 2006). Kolbert's assessment is stark: "It may seem impossible to imagine that a technologically advanced society could choose, in essence, to destroy itself, but that is what we are now in the process of doing" (p. 189). Is it enough now to appreciate being "haunted by waters" or advocate listening to rivers? Must fly fishers and fly fishing seminars become more explicitly political so we have clean, clear, trout-filled waters to ponder and listen to in the future? McGuane (1999) suggests as much in his opening remarks in *The Longest Silence*, proclaiming that "sixty million disorganized fishermen are being hornswaggled by tightly organized and greedy elites…. We really ought to get together" (pp. xiv-xv). In his 1960 "Wilderness Letter" Stegner was hopeful about the transformative impact of nature and wilderness. His "geography of hope" would flow from a rethinking of our relationship to the land. Certainly he contends that we need to stop viewing wilderness as crudely exploitable. Yet he also likely would place fly fishing within the perspective that views rivers and wilderness as resources for "useful" purposes, or as a "playground." He wants us to move beyond such utilitarian ideas as well, in order to celebrate wilderness as an idea, an ideal worthy of preservation and expansion in and of itself—a spiritual resource. But he does recognize that using nature for enjoyment opens us up to the larger existential respect he endorses. Time is limited, though, as he acknowledged in 1980 when he revisited his "Wilderness Letter." In those updated reflections he more pointedly identifies the titanic clash between the public interest and corporate interests as the great crux of the problem. Corporate interests too often triumph at the public's expense. Still, he believes that a profound (and public-regarding) sense of place can be nurtured by close contact with wilderness. That sense of place surely includes waters and fish, a point amplified by Leeson (2009) with reference to the Madison River Valley and Wetherell (2009) in his explorations of Yellowstone Park. The inherent resilience of nature (McKibben 2010; Klinkenborg 2010) also bolsters reason for hope.

The environmental message necessarily is urgent. My seminar is but one small attempt to introduce students to the myriad complexities that arise from engagement with the literature and craft of fly fishing. Thus I offer these observations tentatively. Students (all of us) need to ask deep questions about ourselves and the established practices of our society and our world. After all, as Maclean (1976) put it: "[I]t is not fly fishing if you are not looking for answers to questions" (p. 42). And I suspect Gierach would agree that the lessons learned in the process of searching for answers involve "something we need to know." In closing, may I modestly suggest that we no longer can limit ourselves and our fly fishing students to learning how to read the water. We also need to know how to read ourselves and the world.

Acknowledgements

I would like to thank Will Marquess, Saint Michael's College English Department and Director of the First Year Seminar Program, for his initial encouragement for this project and his untiring support throughout. Doug Facey and Declan McCabe, Saint Michael's College Biology Department, have given freely of their time and energy to make this project work. Dana Baker, Jack Price and especially Rhey Plumley, all of the Central Vermont Chapter of Trout Unlimited, have provided invaluable help with this project, as has The Classic Outfitters fly shop where Rhey works. Jim Millard, Saint Michael's College Senior Instructional Technologist, provided creative technical assistance.

References

Belec, H.M. 2010. TU's Next Generation. Trout 52(3):41-48.

Berners, D.J. 1496. A Treatyse of Fysshynge with an Angle. University of Oregon. Available: https://scholarsbank.uoregon.edu/xmlui/bitstream/handle/1794/680/fishing.pdf?sequence=1. (June 2010).

Crawford, C. 2010. Fly Fishing and First-Year Students. Saint Michael's College Magazine (Winter).

Duncan, D.J. 1983. The River Why. Bantam, New York.

Gierach, J. 1990. "Sex, Death, and Fly-Fishing." Sex, Death, and Fly-Fishing. Simon and Schuster, New York.

Grover, J.Z. 1999. "Why Fish?" Northern Waters. Gray Wolf, St. Paul, Minnesota.

Hemingway, E. (1925). 1996. "Big Two-Hearted River." In Our Time. Scribner, New York.

Hosseini, K. 2003. The Kite Runner. Riverhead Books, New York

Hughes, R. 1999. A Jerk on One End. Ballantine Publishing Group, New York.

Kafka, F. (1915) 2008. Metamorphosis and Other Stories. Penguin, New York.

Klinkenborg, V. 2010. "Public and Private: Restoring a Montana Spring Creek." New York Times (July 9).

Kolbert, E. 2006. Field Notes from a Catastrophe: Man, Nature and Climate Change. Bloomsbury USA, New York.

Lee, R. 1997. Guiding Elliott. The Lyons Press, New York.

Leeson, T. 2009. Inventing Montana: Dispatches from the Madison Valley. Skyhorse Publishing, New York.

Maclean, N. 1976. A River Runs Through It and Other Stories. University of Chicago Press, Chicago.

Martel, Y. 2001. Life of Pi. Harcourt, New York.

McGuane, T. 1999. The Longest Silence: A Life in Fishing. Alfred A. Knopf, New York.

McKibben, B. 2007. Deep Economy: The Wealth of Communities and the Durable Future. Times Books, New York.

McKibben, B. 2010. Eaarth: Making a Life on a Tough New Planet. Times Books, New York.

Quammen, D. 1988. Synecdoche and the Trout. Pages 3-9 in D. Seybold, editor. Seasons of the Angler. Weidenfeld and Nicolson, New York.

Stegner, W. 1960. Wilderness Letter. The Wilderness Society. Available: http://wilderness.org/content/wilderness-letter. (March 2010).

Stegner, W. 1980. Introduction to Wilderness Letter. The Wilderness Society. Available: http://wilderness.org/content/wilderness-letter-introduction. (June 2010).

Walton, I. (1653) 1995. The Compleat Angler. Ecco Press, Hopewell, New Jersey.

Wetherell, W.D. 2009. Yellowstone Autumn: A Season of Discovery in a Wondrous Land. University of Nebraska Press, Lincoln.

Session 3: Genetic Considerations for Managing Wild Trout

PLANNING FOR AN EVOLUTIONARY FUTURE IN THE PRESENCE OF CLIMATE CHANGE: THE QUEST FOR ADAPTIVE GENETIC VARIATION AMONG BROOK TROUT POPULATIONS EXHIBITING PRODIGIOUS DIFFERENTIATION AT NEUTRAL LOCI

Tim King[1]

[1]U.S. Geological Survey, Leetown Science Center, Kearneysville, WV 25430, tlking@usgs.gov

Co-Authors

Moore, S., National Park Service, Great Smoky Mountains National Park, Gatlinburg, TN 37738
Wofford, J. National Park Service, Shenandoah National Park, Luray, VA 22835
Kulp, M., National Park Service, Great Smoky Mountains National Park, Gatlinburg, TN 37738
Morgan II, R., University of Maryland, Appalachian Laboratory, Frostburg, MD 21532
Stauffer Jr., J., Pennsylvania State University, University Park, PA 16802

ABSTRACT — Conservation planning must be framed in terms of providing conditions that will facilitate potential adaptation. To realize this end, resource managers must plan for an evolutionary future for trust species, as such, ecological and evolutionary processes—those that maintain genetic diversity and provide the raw material for evolution and adaptation of populations—must be explicitly identified. Contemporary genomic technology offers great promise for exploring the mechanistic basis of adaptive evolution in a model system. Brook trout *Salvelinus fontinalis* are rich in ecologically and evolutionarily interesting traits (e.g., multiple life history forms; broad latitudinal and elevational distribution; and prodigious gene differentiation (neutral loci) at all spatial scales) that vary between interfertile individuals. Given that both neutral drift and natural selection govern the variance of traits among demographically distinct entities, we are employing a research framework that involves quantifying neutral (i.e., differentiation due to genetic distance) and adaptive genetic variation (measured by mass gene expression profiling) among ecologically and evolutionarily distinct brook trout. As a first step, we have collaborated in an extensive survey of neutral allelic variation at 13 microsatellite DNA loci in over 11,000 brook trout sampled from 275 collections comprising the species' native range. Our traditional population genetic analyses identified prodigous levels of genetic differentiation at all spatial scales. Coalescence-based analyses also illuminated previously undetected demographic histories and evolutionary relationships among populations. We are now in the process of assembling and annotating a transcriptome (*de novo*) and performing mass gene expression profiling via serial analysis of gene expression. We will then remove the gene variation attributable to neutral drift (through phylogenetic subtraction) and the remaining variation significantly associated with ecological factors (e.g., water temperature) can be attributed to adaptive variation. A case study from Great Smoky Mountains National Park and an overall study update was presented.

ASSESSING IMPACTS OF PUT-AND-GROW STOCKING IN SMALL IMPOUNDMENTS ON TRIBUTARY POPULATIONS OF BROOK TROUT IN VIRGINIA

R. Humston[1], N.D. Adkins[1], R.J. Elsey[1], J. Huss[1], B.A. Meekins[1], K.A. Hemminger[1], and T.L. King[2]

[1]Department of Biology, Washington and Lee University, Lexington, VA, 24450. Phone: (540) 458-8341. Email: humstonr@wlu.edu
[2]U.S. Geological Survey, Leetown Science Center, 11649 Leetown Road, Kearneysville, WV 25430

ABSTRACT — Several low-order impoundments in Virginia are stocked annually with fingerling brook trout *Salvelinus fontinalis* in the state's put-and-grow program. We sought to determine if these stocked trout colonize the impoundment feeder streams, and if they naturally reproduce or potentially interbreed with established native populations in these tributaries. Analysis of microsatellite DNA allowed us to distinguish two strains of hatchery-origin trout and one putative native strain among tributaries of stocked reservoirs and one unstocked stream. Hatchery-origin fish were found in tributaries of all stocked reservoirs, mixed with native populations; none were found in an unstocked reference stream that supported wild brook trout. Age-1 trout matching a known hatchery strain were common in tributaries of stocked reservoirs, despite the fact that this hatchery did not supply this year class' cohort of stocked fish, suggesting that stocked trout have successfully reproduced in these streams. Assignment tests identified only three individuals of potentially mixed (native-hatchery) ancestry, all others showed strong affinity to either hatchery or native genotypes. Limited evidence for interbreeding between hatchery-origin and native strains suggests that this practice has had little impact to date on the genetic composition of these native populations. This may be due to assortative mating or other isolating mechanisms between hatchery-origin and native populations.

INTRODUCTION

As the state's only native salmonid, conservation of wild and putative native brook trout *Salvelinus fontinalis* populations is of considerable concern in Virginia. A recent assessment indicated that brook trout have been extirpated from 38% of the sub-watersheds they historically occupied in Virginia, and only 9% of historically occupied watersheds retain 'intact' populations (EBTJV 2006; Hudy et al. 2008). Increased stream temperatures arising from land use practices associated with residential development and agriculture are considered the primary reason for population declines in Virginia; fragmentation of habitats by road building was also noted as contributing to local extirpation (EBTJV 2006). In many areas, inadequate thermal regimes in higher-order streams have caused brook trout populations to persist primarily in headwater habitats. There is currently an increased awareness of land use impacts on water quality in this region due to greater regional and national priority given to restoration of aquatic habitats in Chesapeake Bay and its watershed (Stokstad 2009). Thus, there is a great deal of interest in coldwater fish habitat restoration and re-establishment of brook trout to its native range in the Commonwealth.

Research into the population genetics of extant brook trout populations indicates a high degree of diversity among populations at regional and local scales (Stott et al. 2010), even among populations in close proximity within the same watershed (Richards et al. 2008; King et al. unpublished data). Preserving genetic diversity within and among populations is of considerable importance for conservation strategies (Nelson and Soulé 1987). Genetic diversity is a necessary prerequisite for evolution and adaptation, and preserving diversity enhances the likelihood that populations will persist through disturbance or long-term environmental perturbations (Lande and Shannon 1996). Restoration efforts that potentially reduce genetic diversity, or fail to preserve functional genetic diversity across landscapes (i.e. patterns reflecting local adaption), may inadvertently reduce population resiliency. The common practice of producing fish in hatcheries for release to supplement wild populations can impact population genetics of wild stocks if not properly managed (Marie et al. 2010). Since protocols for stocking practices vary – as do the responses of wild populations to stocking practices – it is important to characterize the effects of stocking on existing populations under a variety of conditions.

In Virginia, several small impoundments are stocked with hatchery-reared brook trout under the state's put-and-grow stocking program. These remote impoundments are fed by low-order mountain streams and were initially established for flood control and municipal water supply. Brook trout fingerlings raised from eggs in the state hatcheries are released directly into the reservoir in the spring (March or April) and grow to large sizes in these deep, cold reservoirs, providing unique (for Virginia) wilderness sport fishing opportunities. Some reservoir tributaries historically harbored wild, putative native brook trout populations. It is unknown to what degree stocked trout colonize these feeder streams and reproduce naturally. If this does occur, introgression between native and hatchery stocks may affect genetic diversity among these extant native populations.

We analyzed microsatellite DNA from tissue samples collected from tributaries of three stocked reservoirs and one unstocked (pure native) stream to determine the following: (1) if hatchery-origin trout disperse into and occupy these feeder streams from the reservoir; (2) if hatchery-origin trout reproduce naturally in these feeder streams; and, (3) if introgression between hatchery strains and native populations has affected the population genetics of these native populations. The tributary of one reservoir did not historically harbor brook trout, allowing us to easily assess colonization in this location. Tributaries of the other two reservoirs had established populations prior to stocking in the reservoirs. Available records indicate that all fish stocked in these reservoirs were reared from a single hatchery strain maintained at the Paint Bank hatchery in Virginia, with one recent exception (G. Duckwall, VA Department of Game and Inland Fisheries, personal communication). In 2008 all stocked fingerlings came from Virginia's Marion Hatchery, which were raised from eggs obtained in 2007 from broodstock of the Rome Lab hatchery strain in New York. Both strains are well established and have been maintained for many generations at their respective hatcheries. We therefore had an opportunity to distinguish the 2008 stocked cohort from natural recruitment arising from spawning of Paint Bank hatchery-origin individuals in the same year.

Methods

In June and July of 2009, we collected fish from Moore Creek, Coles Run, and Mills Creek using a backpack electrofisher (Figure 1). Each of these streams is the only persistent tributary of its respective impoundment. Coles Run and Mills Creek (Potomac River drainage) held wild brook trout populations prior to reservoir stocking, while Moore Creek (James River drainage) did not (L. Mohn, VA DGIF, personal communication; Mohn and Bugas 1980). Annual stocking began in Moore Creek Reservoir in 1984, and in Mills Creek and Coles Run reservoirs in 1989. We also collected trout from Kennedy Creek in June 2009 and 2010; Kennedy Creek is located between Coles Run and Mills Creek in the same sub-watershed of the Potomac drainage. This stream has not been stocked and maintains an established wild trout population; therefore it served as a reference population for the watershed's native trout. These streams are hereafter referred to using descriptive codes identifying their populations as native, stocked, or mixed native-stocked: Kennedy Creek is coded NAT-1; Moore Creek is STO-1; Coles Run and Mills Creek are coded as MIX-1 and MIX-2. We also collected fin clips from broodstock and 2009 year class brook trout from the Paint Bank hatchery to genotype this hatchery strain. Genetic data for the Rome Lab strain had been compiled previously (in 2005 and 2009; King et al. unpublished data), and these data were obtained for comparison.

Fish were measured in the field and tissue samples were collected by snipping a piece of the pelvic fin from each fish. Most fish were released alive; however, in the case of incidental mortality we returned whole fish to the lab for age confirmation from otoliths. Length-frequency data were analyzed to assess the age composition of the collection. The

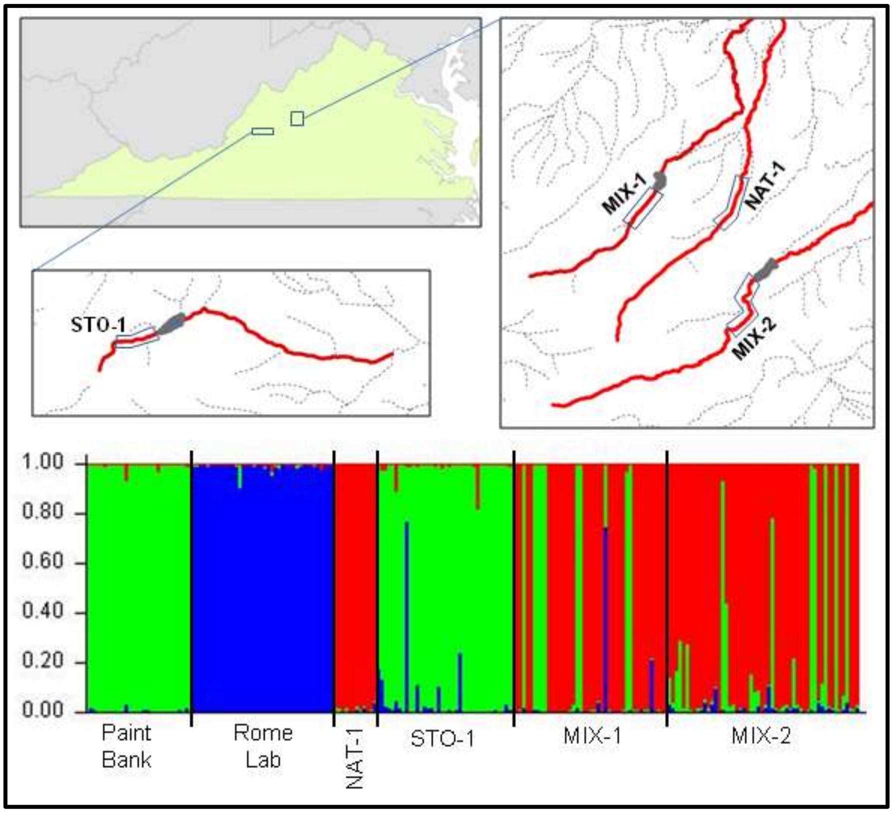


Figure 1. The study streams and their reservoirs are illustrated in the enlarged map areas; polygons highlight the sampled length of each stream. The lower panel illustrates cluster assignments for K=3 populations from Structure. Each individual is represented by a vertical bar with coloration corresponding to its cluster assignment in simulations. Colors correspond to Paint Bank (green), Rome Lab (blue), and native population (red) clusters. Bars with multiple colors stacked represent individuals with varying degrees of ambiguity in their cluster assignment.

package MIXDIST for R statistical software was used to determine the 'mixture' of age-specific size distributions represented in our collection (Macdonald and Du 2010; R Core Development Team 2010). This was useful to assign individuals into year classes for assessing natural reproduction.

Tissue samples were digested and DNA extracted using Qiagen Genomic DNA kits. Samples were genotyped at 10 established microsatellite loci (T.L. King, USGS, unpublished data). We carried out PCR amplification using the Promega M7660 GoTaq PCR Core Kit 2 in a PTC-200 thermal cycler from MJ Research using four different multiplex recipes. Microsatellite loci were analyzed using a LI-COR 4300 DNA analyzer and scored using Saga GT genotyping software (LICOR). All scores were checked for accuracy by eye prior to compilation for data analysis.

Genotypic data were analyzed using a variety of methods to meet the objectives of the study. First, we used MicroChecker software (van Oosterhout et al. 2004) to review the data and test for null alleles. Second, we calculated genetic distance among populations from allele frequencies and visualized relatedness of tributary populations and hatchery strains using unrooted trees. Finally, we used two different methods to assess population structure and assign fish into hatchery or native ancestry groups. The program Structure (Pritchard et al. 2000) was used to determine groupings within our sample based on a Bayesian clustering algorithm with no prior information provided on population membership (i.e. sampling location). The number of subpopulations best supported in the data was determined using the delta-K method described by Evanno et al. (2005). This was necessary because we could not be sure that the genotypes of native populations in MIX-1 or MIX-2 were similar to those from NAT-1. The program GeneClass2 (Piry et al. 2004) was used to assign individuals to either native, Paint Bank Hatchery, or Rome Lab Hatchery ancestry using genotypic data from NAT-1 and hatchery samples as reference data. In cases where assignment results were ambiguous, those individuals were identified as potential native-hatchery 'hybrids', and their genotypes were carefully reviewed.

RESULTS

We collected tissue samples from 140 brook trout from the four study streams and from 39 samples from broodstock and 2009 young-of-year from the Paint Bank hatchery (Table 1). Fish from streams ranged from 41to 245 mm total length (TL) and length-frequency mixture analyses suggested four year classes (2006-2009 YC) were present in the sample (Table 2; Figure 2). The software MIXDIST also estimates the proportion of each age class in the sample, and results indicated that age-1 (2008 YC) were by far the most abundant in the collection (> 60%). Available data from range-wide collections included 200 genotyped individuals of the Rome Lab hatchery strain. We selected a random subsample of 45 individuals from this database for use in subsequent analyses so that sample sizes among populations would not be unbalanced.

Table 1. Sample sizes and size range of fish in collections from study streams and Paint Bank hatchery.

Sample Location	N	Size Range (mm TL)
Moore Creek	39	66-180
Mills Creek	50	47-235
Coles Run	46	46-245
Kennedy Creek	12	41-216
Paint Bank	39	74-321
Total	179	41-321

Table 2: Mean length at age and standard deviation estimated from mixture distribution analysis of length-frequency data. Proportion of each age class in the total sample is estimated as well, and standard error of each estimate is provided where possible.

Age	Year Class	Mean length mm TL (se)	Std Dev (se)	Proportion in Sample (se)
0	2009	53 (1.6)	5.5 (1.3)	0.202 (0.042)
1	2008	101 (2.6)	18.7 (2.7)	0.616 (0.058)
2	2007	177 (8.1)	24.8 (6.8)	0.171 (0.042)
3	2006	240 (NA)	1.9 (NA)	0.011 (0.009)

Results from the program Structure indicated that the data best support separation of three genetically distinct populations within these samples, corresponding to the Rome Lab hatchery strain, the Paint Bank hatchery strain, and a putative native strain (Figure 1). All fish from the Rome Lab and

Paint Bank hatcheries clustered separately with high affinity to their respective groups. All individuals from NAT-1 were assigned to the native cluster. The majority of fish from MIX-1 and MIX-2 were included in the native cluster (79) and a small proportion in the Paint Bank cluster (16). One individual from MIX-1 was assigned to the Rome Lab cluster. All but one individual from STO-1 unambiguously grouped in with the Paint Bank cluster; the remaining individual grouped with the Rome Lab cluster with reasonable confidence. These results support our use of the NAT-1 sample as a reference native strain for this watershed and also demonstrate that the Paint Bank and Rome Lab hatchery strains can be accurately distinguished using these ten microsatellite loci. Individual assignments in GeneClass using NAT-1, Paint Bank, and Rome Lab samples as reference populations were generally consistent with results from Structure. Although we identified five individuals where assignment to native or hatchery origins appeared ambiguous in Structure results (< 80% consistent assignment), posterior probabilities of group membership were generally much higher in GeneClass results (> 95% for three of these five fish). All five of these individuals came from one stream (MIX-2). Structure and GeneClass population assignments were only in disagreement for one individual, and posterior probabilities of group membership were low in both analyses. We closely examined all five of these individuals as possible native-hatchery hybrids.

Assessment of length frequency distributions from each sampling location indicate that several age-1 fish (2008 year class) were of Paint Bank Hatchery ancestry. Most of these came from STO-1, though some were also found in MIX-1 and MIX-2 (Figure 2). Since all hatchery-origin fish stocked in the 2008 year class were of the Rome Lab strain, these fish are most likely the result of natural reproduction by parents of Paint Bank origin stocked in previous years.

Five individuals from MIX-2 were identified as potential native- hatchery hybrids based on assignment results in Structure and GeneClass. We compared their genotypes with allele frequencies from the Paint Bank hatchery and native (NAT-1) populations, purposely looking for group-specific markers. Three of these individuals possessed multiple markers unique to both populations, suggesting interbreeding between hatchery and native strains.

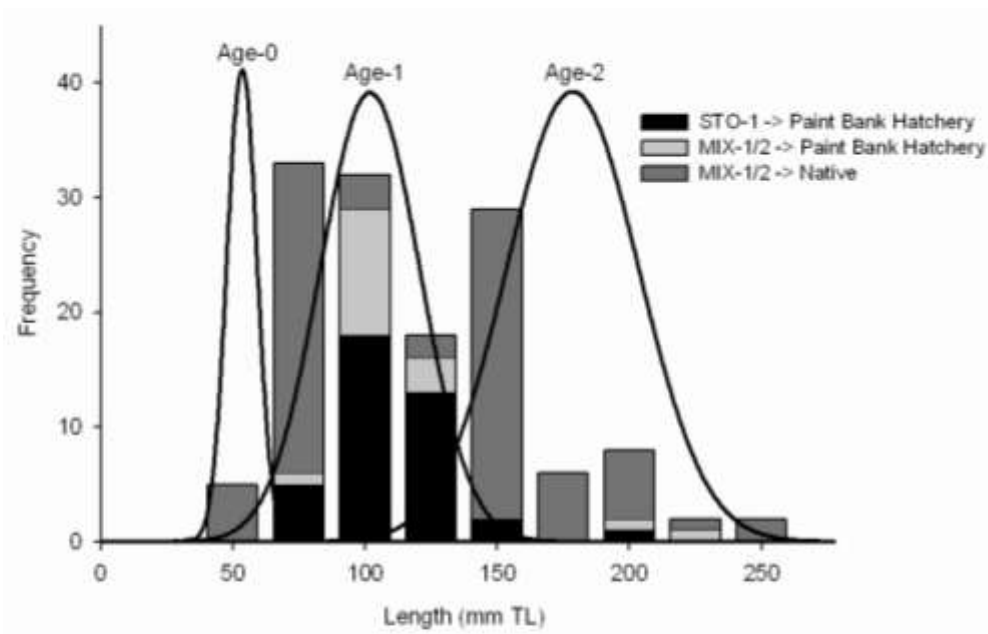


Figure 2. Length frequency distributions of trout collected from the hatchery-stocked reservoir tributary STO-1 and the two tributaries with presumed mixed native-hatchery populations, with groups stacked according to collection stream and genetic ancestry (native vs. Paint Bank hatchery). Superimposed are standardized probability distributions for length at age from mixture distribution analyses. Many age-1 fish matched the Paint Bank hatchery strain despite the fact that this cohort was stocked from a different hatchery strain.

Discussion

Given that we found abundant brook trout in stream STO-1 (previously unoccupied by brook trout) matching the Paint Bank hatchery strain, as well as several fish of Paint Bank or Rome Lab strain in the streams MIX-1 and MIX-2, it is clear that fish stocked in the reservoir disperse into and colonize impoundment tributaries. This is not a surprising result, given documented, widespread dispersal of brook trout following stocking in other areas (Helfrich and Kendall 1982; Dunham et al. 2002; however, see Hudy et al. 2010). In STO-1 brook trout were relatively abundant in the stream, but none were found above a dam created by a felled log in the stream despite the presence of suitable habitat above this dam. This restricted distribution is suggestive of recent colonization. Hatchery strain fish comprised only a small fraction (17%) of collections from mixed native-stocked populations. This could indicate that the presence of a wild population inhibits colonization by stocked fish to some degree. Only two fish in these impoundment tributaries were of Rome Lab ancestry, suggesting that tributary colonization is rare in the first year following stocking. Therefore, subsequent dispersal into tributaries may be related to the onset of reproductive maturity in the second year of life.

The presence of age-1 fish (2008 year class) with genotypes corresponding to the Paint Bank hatchery strain in all three of the impounded tributaries indicates that hatchery fish do successfully spawn after colonizing these feeder streams. We collected young-of-year trout throughout the sampled section of STO-1 in 2009 that were, on average, smaller than the average size of fingerlings stocked in that same year. While this is less conclusive, it further indicates that natural reproduction is occurring in these streams. It remains unclear whether parents of these offspring are tributary residents or lacustrine migrants (with offspring remaining as resident).

Despite apparent naturalization of hatchery strains, we found only limited evidence for interbreeding between hatchery-origin and native fish in these tributaries. The evidence we present for introgression – individuals bearing markers specific to both hatchery and putative native populations – cannot be considered absolutely conclusive for two reasons. First, our sample from the native stream is too small (*n* = 12) to be considered comprehensive – we have augmented this sample with an additional collection (*n* = 25) that is currently being genotyped and will be included in subsequent analyses. It is possible that some of the alleles that are absent from the native sample presented herein will appear in the larger sample. Second, it is likely that the native strains in these neighboring streams are similar but not identical in their allele frequencies. Some alleles absent from the native population in NAT-1 may remain in the native population of either MIX stream population, and these alleles may be shared in common with the Paint Bank hatchery strain. If that is the case, we would mis-identify fish of native ancestry as hatchery-native hybrids. However, assignment results were generally of high certainty and these ambiguous individuals were rare exceptions. Furthermore, these individuals generally had population-specific alleles at 2 to 3 loci from each population. We favor the interpretation that these results support the existence of interbreeding between native and hatchery strains, but (and perhaps more importantly) the extent of interbreeding to date is extremely limited.

It could be argued that there have been a relatively limited number of opportunities for interbreeding between hatchery and native strains; these reservoirs have only been stocked for roughly 20 years. However, it is also possible that assortative mating behavior or asymmetrical competition for spawning habitat or mates limits the potential for interbreeding. Likewise, it is plausible that hatchery strain trout are poor competitors for spawning habitat or breeding females. The limited presence of hatchery fish in the mixed native-stocked tributaries supports both hypotheses to a degree.

Our results suggest that past stocking practices in these reservoirs have not yet compromised the genetic composition of native populations in their tributaries, but continued establishment of a 'naturalized' population of hatchery-origin individuals may eventually lead to introgression even if stocking ceases. In the short term, supplemental stocking with hatchery strains has increased genetic diversity in these systems while preserving putative native genotypes (c.f. Marie et al. 2010). However, the relatively short history of stocking in these systems precludes interpreting this result to mean that supplemental stocking will not lead to loss of native genotypes in the long term. In areas where native populations have been completely extirpated, stocking can help establish new populations in a relatively short period of time.

The future of these impoundments is uncertain; they no longer serve as municipal water supplies, and dam maintenance is limited due to increasing expense as the dams age. There has been discussion of removing these dams and returning the streams to their original free-flowing structure. However, it is unclear what impact this might have on the naturalized hatchery strains in these systems. It is possible that successful spawners migrate into the stream in the fall and return to the reservoirs after spawning. Future research into these populations could seek to identify such potamodromy (lacustrine migrants) as it would impact the persistence of naturalized hatchery populations.

Acknowledgements

We thank Paul Cabe (W&L), Barb Lubinski (USGS), Eric Hallerman (VA Tech), Joanne Davis (VA Tech), and Jamie Roberts (VA Tech) for technical advice and assistance with genetic analyses; Larry Mohn (VDGIF), Paul Bugas (VDGIF), Jason Hallacher (VDGIF), Aaron Coffman(VDGIF), and Dawn Kirk (USFS) for assistance with planning or executing field collections; Bob Carline (Proceedings Editor) for helpful comments on a draft of this manuscript; and the Andrew and Megan Hess Summer Research Scholars program (through W&L) for funding to RH.

References

Dunham, J.B., S.B. Adams, R.E. Schroeter, and D.C. Novinger. 2002. Alien invasions in aquatic ecosystems: Toward an understanding of brook trout invasions and potential impacts on inland cutthroat trout in western North America. Reviews in Fish Biology and Fisheries 12:373-391.

EBTJV (Eastern Brook Trout Joint Venture). 2006. Eastern brook trout: status and threats. Prepared by Trout Unlimited, Arlington, Virginia, for the Eastern Brook Trout Joint Venture. 36 p.

Evanno, G., S. Regnaut, and J. Goudet. 2005. Detecting the number of clusters of individuals using the software STRUCTURE: a simulation study. Molecular Ecology 14:2611-2620.

Helfrich, L.A., and W.T. Kendall. 1982. Movements of Hatchery-reared Rainbow, Brook, and Brown Trout Stocked in a Virginia Mountain Stream. The Progressive Fish-Culturist 44:3-7.

Hudy, M., T.M. Thieling, N. Gillespie and E.P. Smith. 2008. Distribution, status, and land use characteristics of subwatersheds within the native range of brook trout in the eastern United States. North American Journal of Fisheries Management 28: 1069-1085.

Hudy, M., J.A. Coombs, K.H. Nislow, and B.H. Letcher. 2010. Dispersal and within-stream spatial population structure of brook trout revealed by pedigree reconstruction analysis. Transactions of the American Fisheries Society 139:1276-1287.

Lande, R., S. Shannon (1996) The role of genetic variation in adaptation and population persistence in a changing environment. *Evolution* 50:434–437.

Macdonald, P., and with contributions from Juan Du. 2010. mixdist: Finite Mixture Distribution Models. R package version 0.5-3. URL http://CRAN.R-project.org/package=mixdist.

Marie, A.D., L. Bernatchez, and D. Garant. 2010. Loss of genetic integrity correlates with stocking intensity in brook charr (*Salvelinus fontinalis*). Molecular Ecology 19:2025-2037.

Mohn, L.; Bugas, P. E., Jr. 1980. Virginia trout stream and environmental inventory. Federal Aid in Fish Restoration, Project F-32, final report. Richmond, VA: Virginia Department of Game and Inland Fisheries.

Nelson K. & Soulé M. 1987. Genetical conservation of exploited fishes. Pages 345-368 in N. Ryman and F. Utter, editors. Population Genetics and Fishery Management. University of Washington Press, Seattle.

Piry, S., A. Alapetite, J.M. Cornuet, D. Paetkau, L. Baudouin, and A. Estoup. 2004. GENECLASS2: A software for genetic assignment and first-generation migration detection. Journal of Heredity 95(6)536-539.

Pritchard J.K., M. Stephens, and P. Donnelly. 2000. Inference of population structure using multilocus genotype data. Genetics 155: 945–959.

R Development Core Team (2010). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Richards, A.L., T.L. King, B.A. Lubinski, S.E. Moore, L. S. Webb. 2008. Characterization of the genetic structure among brook trout (Salvelinus fontinalis) within LeConte Creek, Tennessee – a recently founded population formed by the introduction of three genetically distinct populations. Proceedings of the 62nd Annual Southeastern Association of Fish and Wildlife Agencies 62:195-202.

Stokstad, E. 2009. Obama moves to revitalize Chesapeake Bay Restoration. Science 324(5931):1138 – 1139

Stott, W., H. Quinlan, O.T. Gorman, and T.L. King. 2010. Genetic structure and diversity among brook trout (*Salvelinus fontinalis*) from Isle Royale, Lake Nipigon, and Lake Superior tributaries. North American Journal of Fisheries Management 30:400-411.

van Oosterhout, C., W.F. Hutchinson, D.P.M. Wills, and P. Shipley. MICRO-CHECKER: software for identifying and correcting genotyping errors in microsatellite data. Molecular Ecology Notes 4:535-538.

Wilson, G.A., and B. Rannala 2003. Bayesian inference of recent migration rates using multilocus genotypes. Genetics 163:1177-1191.

EVOLUTION AND DIVERSITY OF TROUT SPECIES IN THE SIERRA MADRE OCCIDENTAL OF MEXICO

Richard L. Mayden[1]; Casey B. Dillman[1]; Héctor Espinosa-Pérez[2]; Joseph R. Tomelleri[3]; Bernard R. Kuhajda[4]; Dean A. Hendrickson[5]; Gorgonio Ruiz-Campos[6]; Ana Belia de los Santos-Camarillo[1]; Francisco J. García-de León[7]; Alejandro Varela-Romero[8]; David L. Propst[9]; James E. Brooks[10]; Lloyd T. Findley[11]; Anna L. George[12]; David A. Neely[12]; Irene de los Ángeles Barriga-Sosa[13]; Kyle E. Mayden[1]

[1]Department of Biology, 3507 Laclede Ave., Saint Louis University, St. Louis, Missouri 63103 USA.
[2]Instituto de Biología, Universidad Nacional Autónoma de México, México, D.F., México.
[3]Cimarron Trading, 8436 Meadow Lane, Leawood, Kansas 66206 USA.
[4]Department of Biological Sciences, Box 870345, The University of Alabama, Tuscaloosa, Alabama 35487-0345 USA.
[5]University of Texas, Texas Natural Science Center, 10100 Burnet Rd., PRC176 EAST/R4000, Austin, Texas 78758 USA.
[6]Facultad de Ciencias, Universidad Autónoma de Baja California, Apdo. Postal 233, Ensenada, Baja California, 22800, México.
[7]Centro de Investigaciones Biológicas del Noroeste, S.C., Mar Bermejo 195, Col. Playa Palo Santa Rita, Apdo. Postal 128, La Paz, Baja California Sur, 23090, México.
[8]Departamento de Investigaciones Científicas y Tecnológicas, Universidad de Sonora, Apdo. Postal 1819, Hermosillo, Sonora, 83000, México.
[9]Conservation Services Division, New Mexico Department of Game and Fish, P.O. Box 25112, Santa Fe, New México 87504 USA.
[10]U. S. Fish and Wildlife Service, New Mexico Fish and Wildlife Conservation Office, 3800 Commons N.E., Albuquerque, New Mexico 87109 USA.
[11]Centro de Investigación en Alimentación y Desarrollo (CIAD), A.C.--Unidad Guaymas, Carretera al Varadero Nacional km. 6.6, Col. Las Playitas, Guaymas, Sonora, 85480, México .
[12]Tennessee Aquarium, One Broad St., P. O. Box 11048, Chattanooga, Tennessee 37401 USA.
[13]Universidad Metropolitana--Unidad Iztapalapa, México, D.F., 09340 México.

ABSTRACT — The trout species of Mexico's Sierra Madre Occidental (SMO) range have been essentially ignored in science except for the efforts of Needham and Gard (1959) in the mid-20th century. Even after their work documenting the existence of the Mexican Golden Trout *Oncorhynchus chrysogaster* no focused efforts to study the diversity in the SMO south of the USA were initiated until 1997. Sampling in the Sierra Madre Occidental is extremely difficult, but our efforts since that year have revealed that all of Mexico's SMO trout are native (except for hatchery-reared) to the Río Yaqui system southward to the Río Acaponeta, and in the Río Conchos drainage. Morphological and genetic studies of the populations from these rivers support the hypothesis of multiple divergent lineages that we argue are distinct evolutionary species. Conservation and protection of these lineages are critical and should constitute a coordinated effort involving governmental agencies, private organizations, nonprofit groups, and individuals. More inventory work is warranted to better understand the distributions of the native trout and efforts should be made to eliminate the introduction of "hatchery rainbow trout" *Oncorhynchus mykiss* into grow-out facilities in these mountains, instead replaced with propagation efforts on the different native species within their respective drainages. The diversity of wildlife and natural features across the SMO could, with proper planning and maintenance, serve as a fundamental, environmentally sound, sustainable resource for the region via ecotourism.

Mexico's Sierra Madre Occidental (SMO) literally translates as the "Western Mother Range," clearly implying that this great chain of mountains is expansive and especially impressive. These sierras are extremely rugged, less traveled, and, frankly, poorly explored by scientists because of limited road access and activities associated with the cultivation and transport of some illegal plants and their products. It has been considered, and remains today in some areas, a dangerous region for travel and exploration not only because of narcotics activities but also for the unforgiving terrain. Many parts of the range are notorious in some circles of explorers and scientists as only attractive to risk-seeking personalities and the most fit and courageous. The SMO contains some of the deepest ravines, largest canyons, and highest mountains in North America. The vast majority of the area is considered remote and until relatively recently inhabited largely by native tribal peoples, such as the Tarahumara or Rarámuri of Chihuahua, the former Chiricahua Apache in the northern areas and the Northern Tepehuan in southwesternmost Chihuahua and northwesternmost Durango. Spaniards and Anglos almost eliminated the latter two tribal lineages from the region, as well as other lesser-known groups, with invasions and conquests in search of riches and land.

The SMO extends roughly north-south (Fig. 1), from just south of the Sonora-Arizona border southeastward through parts of the states of Sonora, Chihuahua, Sinaloa, Durango, Zacatecas, Nayarit, Jalisco, and Aguascalientes to Guanajuato, where its lower mountain ranges approach the Sierra Madre del Sur and the Eje Volcánico Transversal or the Transverse Volcanic Axis of central Mexico. The Sonoran and Chihuahuan deserts to the northwest and northeast or east, respectively, surround its northern reaches. The range is generally characterized as pine-oak forest or ecological region (not equivalent to the Madrean sky islands of eastern Arizona), and as having high biodiversity and high levels of endemism of specialized species (where studied), particularly several groups of plants (especially pines and oaks) and amphibians and reptiles among animal groups. The evolution of this great biodiversity, much of it unique to this range, has undoubtedly involved major episodic geological events, lack of glacial activities that might drive species to extinction, and long-time variation in micro- and macrohabitats. Isolation by distance, vertically or horizontally, has been a primary driver of speciation and general diversification in the SMO, and ironically, by limiting human access, has been a main reason for the persistence of so many endemic species. The SMO was recently listed as one of six newly designated high-priority biodiversity hotspots worldwide by Conservation International (Mittermeier et al. 2004).

Given the numerous geographically isolated major river systems draining independently into the Gulf of California on the western slope (Fig. 2) (ríos Yaqui, Mayo, Fuerte, Sinaloa, Culiacán, San Lorenzo, Piaxla, Presidio, Baluarte, Acaponeta), and the ríos Conchos (Atlantic, Gulf of Mexico) and Nazas-Aguanaval (endorheic) on the eastern slope, what of the aquatic biodiversity, especially trout, in this area of high species diversity? What of the *truchas nativas*, or native trout, in the headwaters of these large river systems? The fact is, that the combination of incredibly rugged landscape, poor access, and drug-crop cultivation and related activities throughout the region has resulted in a general paucity of knowledge on trout and other aquatic organisms from this part of Mexico. Interestingly, this isolation may have been beneficial for the native trout in many ways. Had they been widely known to anyone other than the indigenous peoples, it is highly likely that their diversity and conservation would have been of limited concern and the various highly distinct gene pools and divergent morphological species in these rivers today would have been "polluted" by USA and Mexican agencies via introduction and transplant programs – paralleling earlier activities that impacted trout diversity and caused the present imperilment of several trout species and populations in the USA. It is a blessing in many ways that as late as 1936, large portions of the Río Yaqui basin had not been substantially modified from a pre-European contact state (Leopold 1949), and this isolation has encompassed many other more southerly river systems harboring native trout. Despite relatively recent and continually increasing construction (destruction) activities in the SMO, bringing new roads and new settlers, travel in most of the region remains extremely difficult, sometimes requiring at least a day of travel by foot or mule to access a single stream. Despite its proximity to the USA in the north and a great deal of recent interest in the terrestrial and aquatic biodiversity of Mexico (e.g., Hulbert and Villalobos-Figueroa 1982; Villalobos-Figueroa 1983; Miller et al. 2005), the aquatic fauna of the SMO has

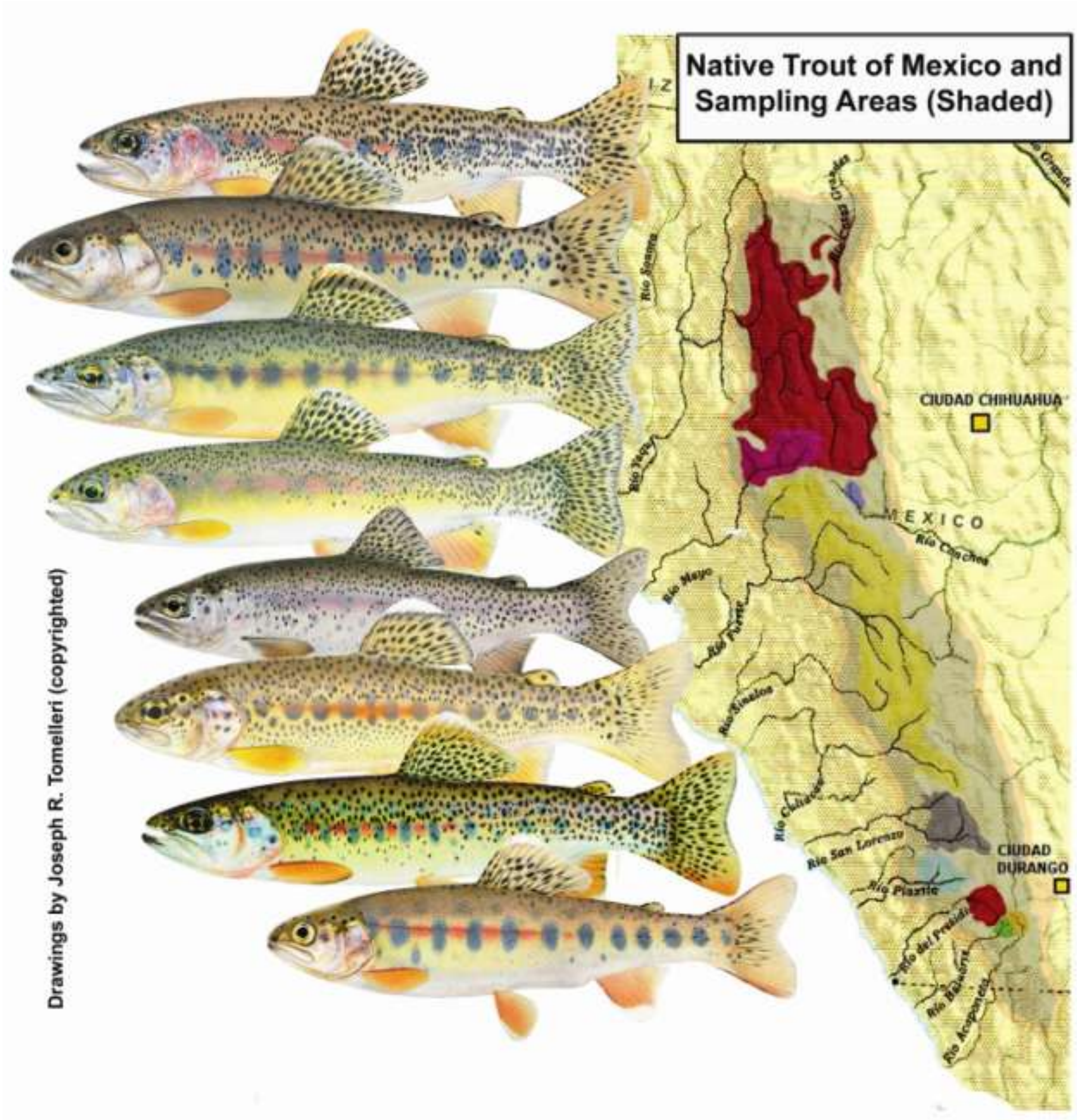


Figure 1. General range of the Sierra Madre Occidental (right) and different undescribed native trout species discovered from the mainland Mexico (left) and their general ranges colored, as well as our sampling region for aquatic biodiversity (right; shadowed area). Illustrations of trout are copyrighted by Joseph R. Tomelleri and used with permission.

remained poorly known. However, with the initiation of a relatively recent and consistent binational effort by a group of biologists, ichthyologists, and citizens from both countries, we now know that trout are much more widespread in the rivers of the SMO than once thought, that they are native, and that they comprise several different lineages considered to be different evolutionary species. Regrettably, these unique fishes are suffering in terms of genetic integrity and conservation status by an increasing onslaught of insults that have paralleled the demise of many of the unique trout lineages previously inhabiting western rivers of the United States and Canada. The time to act to preserve the integrity of these naturally occurring evolutionary lineages, along with their natural habitats and patterns of diversity in the SMO, is now — via coordinated efforts lead by scientists and sociologists and supported by various conservation groups, governmental agencies, public and private organizations and fund raising, along with efforts in ecotourism to demonstrate the value of these fishes to local people, and the continued work of the binational organization formed for the protection of native trout and aquatic habitats of the SMO — *Truchas Mexicanas.* Herein, we provide a brief overview of our findings on native Mexican trout populations in the SMO since 1997.

METHODS

Sampling/Inventory

Likely the most difficult aspects of our long-term project has been the planning (with a fair amount of luck) of survey and sampling expeditions that combine selection of and access to appropriate habitats for native trout, with safety considerations for sampling crews paramount. Most of our efforts have involved hiking and backpacking into remote canyons to streams distant from normally transited roads and with very limited access. Specimens are usually collected by electroshocking gear or angling, maintained alive in stream water, photographed, often weighed and measured at the site, individually sampled for tissues for genetic studies, tagged (labelled) with unique identifiers and preserved as vouchers (specimens) for subsequent morphological study.

Modeling

The SMO encompasses a huge area and, above a certain altitude, watersheds exist that represent potential areas where native trout could occur (Figures 1 and 2). For our early expeditions, sampling sites were selected by (a) where trout had been collected previously in the ríos Yaqui and Mayo systems; or (b) visual inspection of Mexican Government (INEGI) topographic maps and Google Earth projections for likely canyons and rivers or streams meeting our selection criteria. For the most

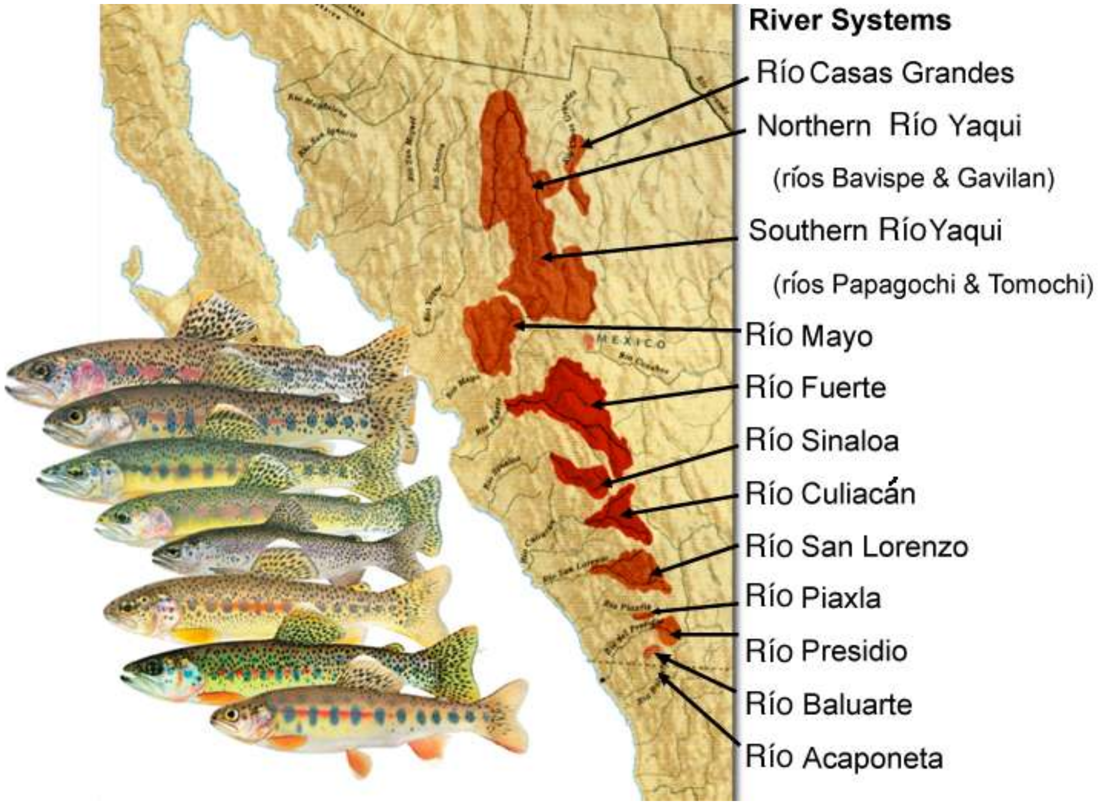


Figure 2. General range of native trout in Mexico and drainages. Illustrations of trout are copyrighted by Joseph R. Tomelleri and used with permission.

part, these approaches proved successful in sampling-site selection. For the Río Conchos basin the approach was augmented by researching pertinent historical literature and archived correspondence dating from the late 1800s and early 1900s by coauthors Tomelleri, Findley, and Hendrickson. These communications clearly indicated that trout had been seen, caught, eaten, and discussed by the indigenous peoples in, and early foreign visitors to, the Río Conchos basin. However, our initial unsuccessful forays to multiple areas of the basin, including several-day excursions into some of its deepest canyons, yielded no trout and called for a more efficient approach to compliment our traditional methods for identifying likely trout habitat and localities. Thus, we began to use a previously tested approach for estimating species' ecological niches, the Genetic Algorithm for Rule-set Prediction (GARP; Stockwell and Peters, 1999; Anderson et al., 2003) to narrow our search for potential trout habitat in the Río Conchos watershed. The GARP algorithm is a machine-learning approach for detecting associations between known occurrences of species and landscape characteristics, particularly where the ecological landscape is complex (such as the SMO) and relationships are not straightforward (Stockwell and Noble, 1992; Stockwell and Peters, 1999). Inspired by models of genetic evolution, GARP models are composed of sets of rules that "evolve" through an iterative process of rule selection, evaluation, subsequent testing, and incorporation or rejection of generated hypotheses (Holland, 1975; Anderson et al., 2003). The GARP program searches for non-random associations between occurrences of species (geo-referenced localities in geographic coordinates) and environmental variables (e.g., digital maps of relevant ecological parameters). We used GARP to identify potential trout habitat in the Río Conchos and other basins, based on environmental parameters derived from remotely-sensed imagery, interpolated station data, and early trout collection localities and habitats in the Río Yaqui basin. The algorithm determines characteristics of a species' habitat based on environmental parameters observed at known collection localities. Similarities across collection locations are used to form conclusions about the distribution of the species based on factors such as temperature, precipitation, elevation, and soil type. Because there is a stochastic element to the GARP process, there is no unique solution. Consequently, a number of models are generated (300 in this case) and a best subset is chosen. The model convergence map was created by the summation of the 10 best-subsets maps. Areas showing a value of 10 reflect highest model convergence. By intersecting the model convergence map with stream coverage of the SMO, streams could be assigned respective model conver-

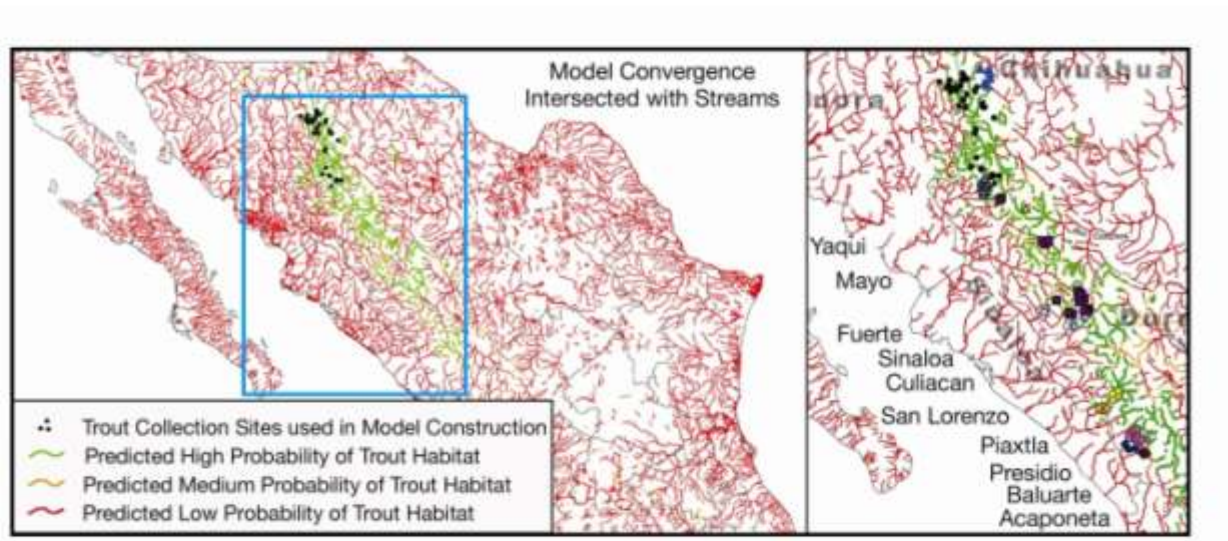


Figure 3. GARP model prediction of distribution of appropriate "niches" for Mexican trout in the SMO, based on information gathered where trout have been sampled in the Yaqui system only (dots). Colored streams on the left indicate relative probabilities of occurrence in a particular stream. Enlarged view of section enclosed in blue box on left is shown on right with sampling locations where we have found native trout, many of the location predictions based on the model illustrated to the left. Note the expansive (green) high probability areas in these rivers and in the ríos Conchos and Nazas-Aguanaval.

gence values (or probabilities of habitat existing that is consistent with what the native trout need for survival = "niche") (Fig. 3). These values can then be ranked to reflect likelihood of the existence, or previous existence, of trout. On the maps presented herein, known localities in the Río Yaqui basin at that time (black dots) were used for modeling. The model correctly identified areas known to contain trout both in the remaining parts of Pacific-slope SMO drainages, as well as predicted trout occurrences in areas of the Río Conchos.

Morphological and Genetic Data and Analyses

Morphological data collection mostly followed those methods outlined by Ruiz-Campos et al. (2003), with the inclusion of additional landmark points for shape analysis. Microsatellite loci, data capture, and analysis mostly followed methods outlined by Nielsen (2001), except that our (ongoing) analyses were performed using a Beckman Automated Sequencer™. Microsatellite data inferences of relatedness between evolutionary lineages involved pair-wise chord distances (D_{CSE}) and neighbor-joining (NJ) analysis. Sequence variation in DNA is measured from the mitochondrial genes control region (1146 bp) and ATPase 6-8 (798 bp) and nuclear genes GH1C and GH2C; sequences are aligned and analyzed using maximum parsimony, maximum likelihood, and Bayesian analyses using as outgroups other (non-trout) described species of *Oncorhynchus* (salmons), as well as *Dalia*, *Esox*, *Salmo*, and *Salvelinus*.

Diversity of Trout Species in the SMO

Information presented below is derived from ongoing work and, in part, from Hendrickson et al. (2003, 2006). Here, we present a brief overview of what we have come to hypothesize regarding the diversity of evolutionary lineages of trout species in Mexico and their phylogenetic relationships (all of these distinct lineages are part of the larger "rainbow trout" clade and not the "cutthroat trout" clade.

Prior to 1997, SMO trout specimens (*Oncorhynchus* spp.) available for scientific study in museums came from a limited number of localities in the drainages of the ríos Yaqui, Mayo, Fuerte, Sinaloa, and Culiacán; the latter three harboring what Needham and Gard (1959) named and considered the Mexican Golden Trout, *O. chrysogaster*. Whether trout from the ríos Yaqui and Mayo drainages were native or exotic was unknown as their taxonomic status had been ignored. Trout populations in rivers south of the putative range of *O. chrysogaster* had been linked with unfounded information ("legends") claiming that they were simply introduced and escaped hatchery trout, *O. mykiss*. In general, knowledge of the distribution and diversity of native trout in Mexico has been in a state of chaos (see Figs. 1-3). Conducting field studies in the habitats where they exist, however, can be extremely difficult. However, our modeling efforts have aided our efforts tremendously and have correctly identified areas known to contain trout, both in the remaining parts of Pacific-slope SMO drainages, as well as predicted trout occurrences in areas of the Río Conchos drainage (Fig. 3).

Since 1997, our multiple sampling efforts have discovered self-sustaining populations of native trout in high elevation streams from the ríos (north to south) Yaqui, Mayo, Fuerte, Sinaloa, Culiacán, Presidio, San Lorenzo, Piaxla, Baluarte and Acaponeta, and the Río Conchos (Figs. 1-3). Independently, Ruiz-Campos et al. (2003) initiated a single-season sampling effort in many of these drainages and documented the existence of native trout as well as previously undescribed diversification. As previously noted, other than *O. chrysogaster* from three mid-SMO drainages, little was known or published on the taxonomic status or origins of other native trout. We do know, however, that for many years Mexican government agencies have trucked larval hatchery trout (*O. mykiss*) to what has been estimated as "hundreds of 'grow-out' facilities" (of varied quality) to mature and then provide to local people. Robert R. Miller (pers. comm.) and Miller et al. (2005) argued that all trout south of the putative range of *O. chrysogaster* were the result of in-stream introductions or hatchery escapees, an argument that we now know is wrong.

Variation in morphological and DNA microsatellite-sequence data of the multiple populations from the Río Yaqui southward to the ríos Baluarte and Acaponeta, as well as two widely disjunct populations in the Río Conchos basin, clearly support the existence of multiple independent evolutionary lineages (or new species) in the Pacific-slope drainages mentioned above, plus two separate headwater areas in the Atlantic-slope Río Conchos drainage. The phylogenetic relationships of these

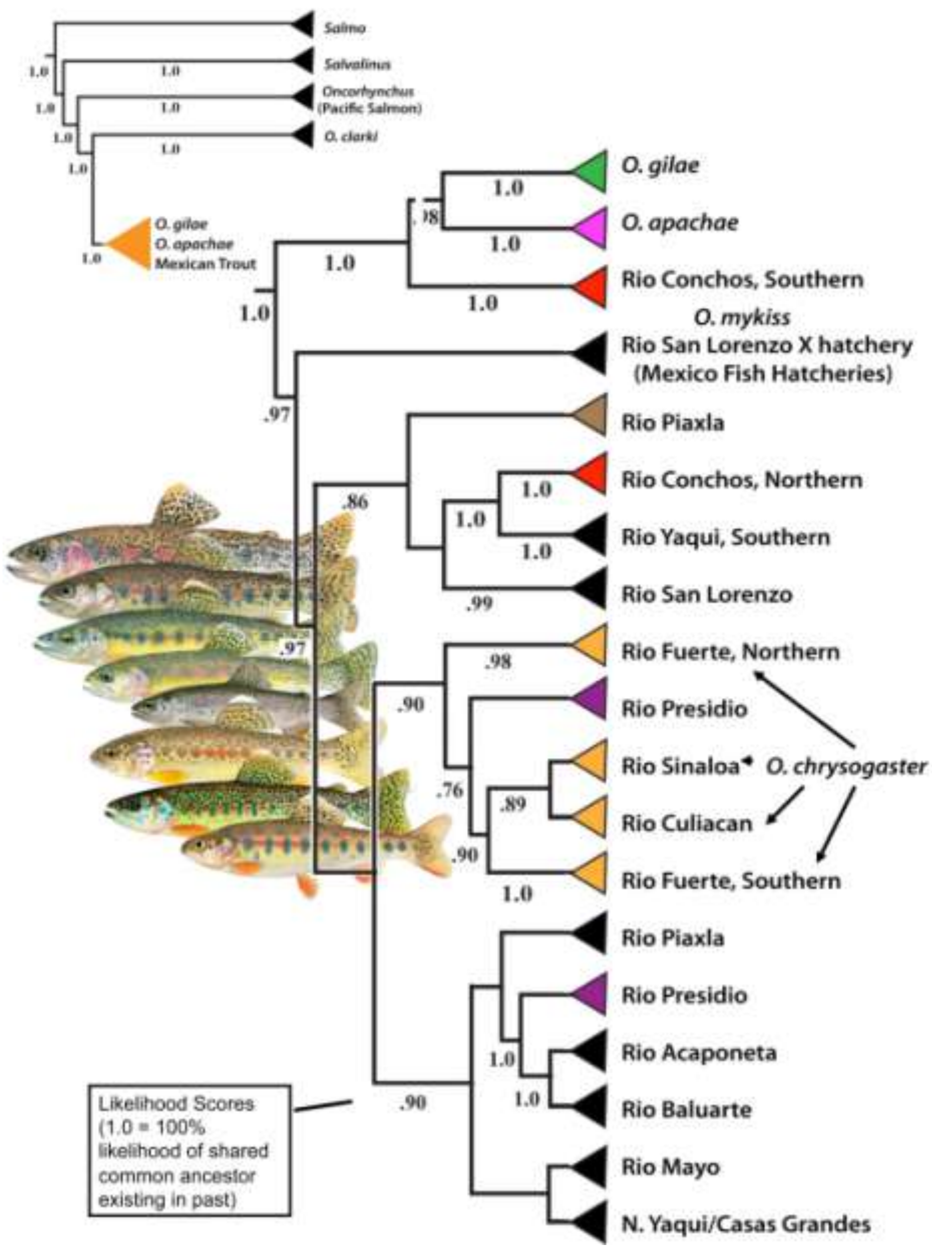


Figure 4. Maximum likelihood analysis of the phylogenetic relationships of Mexican trout species, inclusive of the Mexican Golden Trout, *O. chrysogaster*, and multiple new species being described. Upper left illustrates basal relationships of trout and salmon and the sister group relationship between the "cutthroat" and "rainbow" lineages, the latter of which the Mexican species and *O. apache* and *O. gilae* are descendants and form a monophyletic group (expanded tree). The expanded tree illustrates the phylogenetic relationships and naturalness of the different lineages restricted to particular river basins in Mexico. The naturalness of these species lineages is concordant with results from parallel microsatellite analyses. Note that all of the hatchery rainbow x native trout hybrids all fall out in a single lineage. Illustrations of trout are copyrighted by Joseph R. Tomelleri and used with permission.

divergent lineages are presented in Figure 4, where it is seen that some lineages are more closely related to the described *O. chrysogaster* (and the O. apache plus O. gilae lineage) than to other undescribed xlineages in the SMO. Of considerable interest is that the southernmost undescribed species from the Río Conchos drainage is sister to the *O. gilae / O. apache* clade. Among other intriguing observations of these relationships and their geographic locations is that there may exist two distinct forms in each of the ríos Piaxla and San Lorenzo. Juveniles have been collected in both rivers at times indicating multiple spawning episodes during a year. Furthermore, in some other southern drainages, some collections produced smaller, more brilliantly colored adults with ripe ova and running milt as well as elongated, muted-color larger specimens not in breeding condition. These observations might support the hypothesis that possible multiple "run-time" forms (or species) exist in these southern drainages. This hypothesis requires much more extensive sampling and field study to adequately test such putative diversity.

Impacts of Hatchery-reared Trout and Grow-out Facilities

One major concern of biologists studying naturally existing trout populations in the USA and Canada is how hatchery-reared trout have impacted native populations in different streams. The same concern is another focus of our research because sampling efforts (voucher specimens and tissues) include both native and exotic hatchery trout from streams near grow-out facilities, as well as specimens sampled directly from those facilities. Objectives in this regard are to determine extent of hybridization of hatchery-reared (rainbow) trout with native trout and at what distance from grow-out facilities are alleles characteristic of hatchery-reared trout found? To date, microsatellite analyses of 13 loci have identified some hybridization, especially in close proximity to grow-out facilities, but introgression with hatchery trout is (luckily) not yet widespread. An important aspect of the results of the microsatellite analyses is that trout populations from rivers and tributaries of the Río Yaqui system, in northern Chihuahua and Sonora, southward to the ríos Baluarte and Acaponeta are not hatchery-reared strains or introduced fish. They instead represent divergent native lineages of trout, corroborated by morphological and genetic data that are part of a larger monophyletic group that we term the "Rainbow Trout" clade.

Conservation Status

While our efforts have involved a large component of sampling and biologically analyzing native trout populations from the many headwaters on both sides of the continental divide of the SMO, we have been unable to expend a great deal of time assessing the different lineages for their current conservation status. Two exceptions to this general observation are the distinct lineages from the northern and southern parts of the divergent Río Conchos drainage. Distributions of both of these undescribed forms are very restricted, limited to a few stream-kilometers, and population numbers are quite low at other locations. Near the current range of the Northern Conchos Trout is a semi-abandoned grow-out facility for hatchery trout, but sampling inside its concrete enclosure did not yield any hatchery trout. However, this facility could be rapidly reactivated and no barriers exist to prevent escapees from moving upstream into the only stream documented to support native Northern Conchos Trout. This as-yet-undescribed species, as well as others, has been subjected to local fishing pressures involving poisoning of individuals (seeking cover under in-stream boulders) with certain native plant extracts or commercial bleach. All dead or stunned fishes are then retrieved by moving the boulders with large tree trunks used as levers. While we have no data to quantify the extent of this method, our clear impression is that, since our initial sampling began, this type of unsound fishing activity has increased across the SMO. Likewise, large parts of the Río Conchos drainage have been heavily tapped for irrigation purposes while also undergoing extended drought, factors which almost certainly are exacerbating pressures that are normally exerted by an agrarian population larger than the region can sustain, even under more mesic conditions. The Southern Conchos Trout is restricted to a very short segment of a single stream, portions of which are only spring-fed, but streamflow and trout numbers could vary dramatically, making it (as with the Northern Conchos Trout), very vulnerable to rapid extinction.

In general, all extant trout populations in the SMO exist in areas impacted by timber harvest, especially new or increasing harvest, and are thus especially vulnerable to rapid decline, extirpation, or extinction as a consequence of habitat degradation. Some timber harvest methods (luckily) have been highly selective and cause limited habitat modification, whereas in several other areas large tracts have been negatively and extremely impacted by extensive loss of forest cover (including the southern branch of the Río Conchos). As documented for other ecosystems where large tracts of forested or generally vegetated watersheds are partially or completely denuded, streams become heavily impacted by siltation and increased water temperatures. Accumulated silt compromises spawning habitats for fishes and impairs development of embryos requiring a clean, percolating and circulating water supply within the substrate. Siltation also eliminates habitats for many micro- and macroinvertebrates that serve as a food source for trout. Salmonids are coldwater species highly sensitive to increased water temperatures accompanying loss of riparian cover. Expansion of timber harvesting and creation of logging roads into SMO forests (containing pristine trout habitat) will promote human population influx and more settlements, agriculture and livestock, all leading to the demise of these species via increased siltation, overfishing, and pollution of local water supplies, if not planned properly. In the coming years, plans must be developed or expanded by conservation entities that are supported by the government or private endowments for the establishment and management of preserves to maintain the existing trout diversity in the SMO. An excellent example of this is the activity supported by World Wildlife Fund (WWF)-Mexico's Chihuahua branch, for the area occupied by the Northern Conchos Trout.

Positive Potential for Ecotourism in the Region

The areas of the SMO supporting native trout and intact or nearly-intact forests offer great potential for pubic appreciation of scenic landscape, natural diversity, emotional tranquility, and enjoyment of the natural world. Such activities need not exclude trekking, climbing, rafting, swimming, camping, cabañas in relatively isolated landscapes, etc., and could include guided trout fishing excursions extending for hours to days. Local residents already knowledgeable of the terrain, trained in medical aid, and with proper equipment could hire out to people from all over the world who are known to be interested in fishing for new (and beautiful) trout species that only a few nonlocals have ever seen in the wild. Such programs would need coordination with plans for the establishment and management of preserves, catch (creel) limits, and potentially only catch-and-release fishing in some streams. A multitude of collateral activities could be established in these new and sustainable developments to promote local cultural appreciation, education and experiences, responsible resource utilization, and many other environmentally proactive activities, while one or more members of a family enjoys trout fishing.

Projected Needs and Priorities for Sustainable Trout Diversity in Mexico

The long-term existence of this great diversity of trout in steams of the SMO, with its large tracts of still intact or nearly intact forested areas, will be impossible without urgent proactive measures by committed people and organizations. Financial support is not the only answer to this dilemma, as has been clearly demonstrated historically and repeatedly by management and protection efforts of native trout species, subspecies, and populations in the United States by local, regional, and national organizations. Additionally, some negative trends have been reversed via an activity initiated by *Trout Unlimited*, more than a decade ago, entitled *Bring Back the Natives*. After decades of fisheries management techniques that focused on stocking hatchery trout into streams and moving trout species around the United States, it finally occurred to some that species really evolved in, and thus are adapted to, particular environmental regimes, and that everyone would be better served if focus were instead placed on trout native to particular drainages and watersheds, and their potential propagation and supplementation for conservation and angling purposes. Such an organized, bottom-up effort may likely be the best approach for Mexico to conserve its splendid diversity of native trout, rather than repeating mistakes previously made in the USA.

Governance, Oversight, and Priorities

Such an effort will require a committed governance board of highly motivated persons with clear and demonstrated priorities in the conservation of natural diversity. The board would serve many functions, as determined by the eventual organization, but it would have to be focused on necessities of preserving the SMO's natural aquatic biodiversity. It would represent and "speak for" the best interests of the natural diversity with many different agencies, organizations, and cultural and industrial entities, all potentially competing with one another and the natural diversity for the same resources. Such a governance body would have to contain members knowledgeable in broad and important topics, such as knowing the resources and the literature -- while holding primary the interests of the aquatic diversity. It would be highly recommended that Mexican and international organizations, including those concerned with wild salmonids, be involved as much as possible in building a strong base of support, not only for the protection of the diversity but also for fund raising to drive such an initiative.

Natural Diversity

The objective of such an organization would be to maintain the existing natural diversity of known trout populations and their ecosystems, as well as overseeing continuing survey/inventory efforts that must be conducted across other areas of the SMO that likely harbor as yet unknown diversity potentially in need of description, protection, and management.

Aquaculture and Management of Stocking Efforts

Aside from habitat deterioration and overfishing, the spread of alien (hatchery) trout represents the most powerful and greatest threat to the natural diversity of trout species in Mexico. It is likely that practices addressing this large problem will be modeled after programs conducted in other countries for the management of trout in the wild and in hatcheries, and possibly some pertaining to other (somewhat irrelevant) species such as tilapias and steelhead. In recent years, the Mexican government has become increasingly interested in the natural diversity of its trout populations and in limiting importation of hatchery (rainbow) trout eggs and larvae in favor of aquaculture practices informed by the diversity and geography of native species. Recently, experimental studies conducted at the Guachochi hatchery in southern Chihuahua focused on rearing young native trout to adult stages and conducting selective crosses. Results have been very positive in reaching competitive sizes (compared to hatchery trout) and in survival of fry. Such efforts should be continued (and expanded), with strict monitoring for conservation purposes and for those interested in angling for large-trout species. Local hatchery production of native species should supply grow-out facilities only within the watersheds of those same species. Hatchery trout (*O. mykiss* varieties) should be stocked only (if at all) in facilities located within river basins containing no native trout, and every effort must be made to retain those fish within such facilities. Escaped hatchery trout will destroy native ecosystems by consuming and competing with other fishes, macroinvertebrates, and amphibians that are essential in the food webs of streams.

Funding and Not-for-profit Fund Raising and Contributions – Private, Agency, and Corporate Sponsorship

Today, discovery and protection of natural diversity requires financial support for a whole variety of activities. Now that we have a relatively firm understanding of the diversity of trout in Mexico, funding will be absolutely necessary for population assessments, conservation genetics, impacts of hatchery trout on the genetics of native populations, acquisition of land and water rights to protect high-priority habitats, and to educate the public as to why such activities are essential for not only their generation but future generations as well. Thus, we strongly urge that every effort be put forth by established conservation organizations, via and including individual citizens, for targeted fund raising to achieve such much needed primary goals: Conservation, responsible trout propagation, and repatriation of the two Río Conchos trout species, as well as conservation genetics, and additional survey andinventory efforts across the vast areas of prime native trout habitat in the SMO. It is critical to emphasize that conservation and management issues for native SMO trout should not be addressed with only money and private funding, or only one agency in Mexico. Success will come only with support from all pertinent Mexican federal agencies (e.g., SEMARNAT, SAGARPA-CONAPESCA, CONABIO, etc.) working toward an integral national plan driven by the strategies of governance, oversight and priorities.

Acknowledgments

This long-term research effort would have been impossible without financial, logistic and other support from several Mexican institutions and organizations, including Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, Universidad Nacional Autónoma de Mexico, Universidad Autónoma de Baja California, Universidad de Sonora, Centro de Investigaciones Biológicas del Noroeste, Centro de Investigación en Alimentación y Desarrollo, Universidad Autónoma Metropolitana-Iztapalapa, Secretaría de Agricultura, Ganadería, Desarrollo Rural, Pesca y Alimentación, Comisión Nacional de Ciencia y Tecnología, and World Wildlife Fund-Mexico. Also, USA National Science Foundation (DEB 0817027 to RLM), Saint Louis University, University of Texas, US Fish and Wildlife Service, New Mexico Department of Game and Fish, and (very often) the personal funds of many of the participants involved in exploration and inventory, and genetic and morphological studies of the native trout populations of the Sierra Madre Occidental.

Literature Cited

Anderson, R.P., D. Lew, and A.T. Peterson. 2003. Evaluating predictive models of species' distributions: criteria for selecting optimal models. Ecological Modelling, 162:211-232.

Camarena-Rosales, F., G. Ruiz-Campos, R. L. Mayden, D. A. Hendrickson, A. Varela-Romero, and F. J. García de León. 2007. Mitochondrial haplotype variation in wild trout populations (Teleostei: Salmonidae) from northwestern Mexico. Reviews Fish Biology and Fisheries 18:33-45.

Hendrickson D. A., H. Espinosa-Pérez, L. T. Findley, W. Forbes, J. R. Tomelleri, R. L. Mayden, J. L. Nielsen, B. Jensen, G. Ruiz-Campos, A. Varela-Romero, A. M. Van Der Heiden, F. Camarena, and F. J. García de León. 2003. Mexican native trouts: a review of their history and current systematic and conservation status. Reviews Fish Biology and Fisheries 12:273-316.

Hendrickson, D. A., D. A. Neely, R. L. Mayden, J. E. Brooks, F. Camarena Rosales, R. Cutter, H. Espinosa, L. T, Findley, F. J. Garcia de León, B. R. Kuhajda, D. L. Propst, G. Ruiz Campos, E. St. Clair, J. R. Tomelleri, and A. Varela Romero. 2006. Conservation of Mexican native trouts and a white paper on the discovery, status, protection, and recovery of the Conchos Trout, the first native *Oncorhynchus* of the Atlantic Drainage in North America, p. 162-201. In: Ma. De Lourdes Lozano-Vilano and Armando J. Contreras-Balderas (Eds.). Studies of North American Desert Fishes in Honor of E. P. Pister, Conservationist. 652 pp. Universidad Autónoma de Nuevo León, Mexico.

Leopold, A. 1949. A Sand County Almanac. Oxford University Press, New York.

Miller, R. R., W. L. Minckley, and S. M. Norris. 2005. Freshwater Fishes of Mexico. University of Chicago Press. 652 pp.

Mittermeier, R.A., G. A. B. da Fonseca, M. Hoffman, J. Pilgrim,T. Brook, Robles Gil, P., Mittermeier, C.G. and Lamoreux, J. 2004. Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions. CEMEX, Conservation International, Agrupación Sierra Madre, University of Virginia, Washington, D.C.

Needham, P. R., and R. Gard. 1959. Rainbow trout in Mexico and California with notes on the cutthroat series. University of California Publications in Zoology 67:1-124.

Nielsen, J. L., M. C. Fountain, J. C. Favela, K. Cobble, and B. L. Jensen. 2001. *Oncorhynchus* at the southern extent of their range: a study of mtDNA control–region sequence with special reference to an undescribed subspecies of *O. mykiss* from Mexico. Environmental Biology of Fishes 51:7-23.

Peterson, A.T., and D.A. Vieglais. 2001. Predicting species invasions using ecological niche modeling: new approaches from bioinformatics attack a pressing problem. BioScience 51:363-371.

Ruiz-Campos, G., F. Camarena-Rosales, A. Varela-Romero, S. Sánchez-Gonzáls, R. De La Rosa-Vélez. 2003. Morphometric variation of wild trout populations from northwestern Mexico (Pisces: Salmonidae). Reviews Fish Biology and Fisheries 13:91-110.

Stockwell, D., and D. Peters. 1999. The GARP modelling system: problems and solutions to automated spatial prediction. International Journal of Geographical Information Science 13:143-158.

Stockwell, D. R. B. and I. R. Noble. 1992. Induction of sets of rules from animal distribution data: a robust and informative method of analysis. Mathematics and Computers in Simulation 33:385-390

Recovery Planning for Westslope Cutthroat Trout: A Threatened Species in Alberta

Jennifer E. Earle
Fisheries Biologist, Fish and Wildlife Division, Alberta Sustainable Resource Development, Box 1420, Cochrane, Alberta T4C 1B4, Canada. Corresponding author: Jennifer.Earle@gov.ab.ca

ABSTRACT — In Canada, westslope cutthroat trout *Oncorhynchus clarkii lewisi* were assessed by federal and provincial species-at-risk bodies, which recommended a Threatened status for the Alberta population. In Alberta, the species was legally listed under the Wildlife Act in fall 2009, with the requirement that a recovery plan be completed within 2 years. The Alberta population of westslope cutthroat trout is currently being considered for listing pursuant to *Canada's Species at Risk Act*. A joint federal-provincial recovery team was established in January 2009, prior to the actual listing, and consists of representatives from provincial and federal management agencies, conservation groups, industry and universities. One of the most problematic issues the team has faced is with identifying "pure" westslope cutthroat trout populations. The issues are complicated due to historic and widespread introductions of rainbow trout *Oncorhynchus mykiss*, which has resulted in hybridization, as well as the stocking of westslope cutthroat trout both inside and outside of native range. The metric *q*wsct was used to describe the genetic purity of populations, which represents the proportion of each fish's genome that is attributable to westslope cutthroat trout. A pure population is considered one in which the average *q*wsct-value is ≥0.99. At this stringent level, however, the number of populations in the province which are considered pure is extremely low with many of these remnant populations restricted to headwater areas, frequently above impassable barriers. The recovery team's current direction is to recommend prioritization of native populations based on genetic status. The reality is that for many of these populations, maintenance will become the recovery goal, due to their small size and isolated nature. While the emphasis is on protecting the native pure populations, the team acknowledges the importance of introduced pure populations, as well as those exhibiting slight hybridization.

Introduction

Historically one of the first fish to recolonize western Canada following glaciation, cutthroat trout *Oncorhynchus clarkii* are often the only native trout species in many locations (ASRD and ACA 2006). Fourteen subspecies of cutthroat trout are generally recognized (Allendorf and Leary 1988; Behnke 2002). Of these, four major subspecies (coastal *O. c. clarkii*, westslope *O. c. lewisi*, Lahontan *O. c. henshawi* and Yellowstone cutthroat trout *O. c. bouvieri*) are widely distributed and show considerable divergence from each other. Populations of westslope cutthroat trout in British Columbia and Alberta exist on the northern periphery of the subspecies' range, which extends over both sides of the Continental Divide as far south as Montana, Idaho, Washington, Oregon and Wyoming (Figure 1). Although historically a widespread species, westslope cutthroat trout has shown dramatic global declines in the number and distribution of populations. Habitat degradation, overharvesting and the introduction of non-native species (through increased competition and hybridization) have all contributed to the decline (ASRD and ACA 2006).

Figure 1. Original global distribution of westslope cutthroat trout (shaded). Modified from ASRD & ACA (2006). Distribution data primarily from Behnke (1992), also see Mayhood (2009).

Federally, the national status of westslope cutthroat trout was reviewed by the Committee on the Status of Endangered Wildlife in Canada in November 2006 (COSEWIC 2006a, 2006b). Two designatable units for the species were created at the time, consisting of one unit in British Columbia, and one unit in Alberta. A determination was made based upon the marked difference in conservation status and distinctive ecozones inhabited by the two groups and the lack of current dispersal opportunities between them (separated by the Rocky Mountains). The population in British Columbia is listed *Special Concern* due to habitat loss and degradation, as well as competition and hybridization with introduced species. In Alberta, the species is designated as *Threatened* for the reasons outlined below. It should be noted that the assessment included only genetically pure, unstocked populations of the species occurring within their historical range.

In December 2007, Alberta's Minister of Sustainable Resource Development approved listing the westslope cutthroat trout as *Threatened* under Alberta's *Wildlife Act,* based on the recommendations from the Endangered Species Conservation Committee (ESCC). This designation was due to the subspecies' small distribution and continuing decline in extent of occurrence, the severely fragmented nature of populations, continuing decline in quality of habitat, and dispersal barriers making immigration between watersheds, and therefore rescue of the Alberta population from other jurisdictions, highly unlikely.

The species was formally listed in the regulations under the Alberta *Wildlife Act* in fall 2009, with the requirement that a recovery plan be completed within 2 years. A recovery team was assembled in January 2009, prior to the actual listing, and consists of representatives from provincial and federal management agencies, conservation groups, industry and universities. At the time of writing, the decision to list the species under federal Species at Risk legislation is pending.

Project Background

Since 2006, the province has conducted or supported fieldwork in areas known or suspected of containing pure westslope cutthroat trout. The main data priorities have been to identify remaining genetically pure, unstocked populations; determine their distribution, abundance and size structure; as well as identify barriers to upstream migration of non-native species.

A significant component of the project has been to conduct genetic analyses to investigate the extent of (i) hybridization and introgression between westslope cutthroat trout and rainbow trout (*O. mykiss*) and (ii) population subdivision among pure westslope cutthroat trout populations.

Although significant work has been conducted in the upper Bow River drainage, within Banff National Park, this paper will concentrate on work conducted in provincially managed waters.

Study Area

Historically in Alberta, westslope cutthroat trout occupied the Bow and Oldman river drainages and accessible tributaries (i.e., below waterfalls and other impassable barriers), and may have extended downstream into the upper Milk River drainage of Alberta from the Montana headwaters (Prince et al. 1912; Behnke 1992). Numerous historical records indicate that these trout were abundant throughout most of the native range in Alberta (Mayhood et al. 1997 and Mayhood unpublished data).

At present, native westslope cutthroat trout occupy considerably less than 5% of the native range in the Bow drainage, where they appear to be restricted to the extreme headwaters of a few of the major tributaries and the upper main stem (Mayhood 1995, 2000; ASRD 2008). Many remaining Bow drainage populations within the native range appear to be, or are known to be, hybridized (McAllister et al. 1981; Strobeck 1994; Bernatchez 1999; Carl and Stelfox 1989; Janowicz 2005; Taylor and Gow 2007; Robinson 2008; ASRD 2008); nearly all remnant populations are small and isolated (Mayhood 2000; ASRD 2008).

In the Oldman River drainage, westslope cutthroat trout still occupy most of the native range in the upper Oldman basin, but have been lost from native waters in the mainstem east of the mountain front and most of its fish-accessible tributaries (Radford 1975, 1977; Fitch 1977–80; Mayhood et al. 1997). Westslope cutthroat trout are uncommon to rare in the St. Mary and Belly drainages (including the Waterton drainage) and have been all but extirpated from their native waters in the Crowsnest drainage (Fitch 1977–80; Mayhood et al. 1997; ASRD 2008).

Methods

Genetic analyses and reporting was conducted at the University of British Columbia, Department of Zoology, Biodiversity Research Centre and Native Fishes Research Group under the direction of Dr. Eric B. Taylor. Allelic variation at nine microsatellite DNA loci was assayed in trout *Oncorhynchus* spp.. Tissue samples were fin clips stored in 95% ethanol or dried in scale envelopes. In general, a minimum of 30 to 35 tissue samples were collected from each site sampled; however, at some sites densities were extremely low and fewer samples may have been collected.

The genetics data were summarized by reporting the “westslope cutthroat trout ancestry coefficient (*q*wsct)” for each fish and the average across fish for each locality. The *q*wsct is an index of the proportion of each fish’s genome that originates from westslope cutthroat trout (e.g., “pure” westslope cutthroat trout would have a *q*wsct = 1.0, F1 hybrids a value of 0.5 and pure rainbow trout a value of 0). The value of *q*wsct was used because it should also be more directly comparable with various thresholds that have been suggested (e.g., 0.99 westslope cutthroat trout for a fish to be considered “pure”, Allendorf et al. 2004). For a complete description of genetic analyses methods, refer to Taylor and Gow (2007, 2009).

Field sampling was generally conducted using a backpack electrofisher (Smith-Root Model LR-24); some samples were also collected by angling. Barrier surveys were conducted both aerially by helicopter and by foot, depending on site conditions.

Results

Genetic Analyses

Between 2006 and 2009, approximately 300 sites were surveyed and over 3,500 tissue samples were collected. A coarse level roll-up of the genetic data indicated that roughly 33% of the populations sampled were pure on average at the 0.99 level (Figure 2). A more detailed breakdown of the data by drainage indicated that pure populations were well distributed at sites in the Oldman River drainage, but were confined to a small number of sub-basins in the Bow River drainage, especially in the Highwood River drainage. Pure rainbow trout were generally found in tributaries to lower reaches of rivers and often below impassable barriers. The corollary to this was that pure populations of westslope cutthroat trout were usually located in the upper reaches of streams and often above impassable barriers, such as waterfalls or extensive sections of subsurface flow.

In populations where hybrids were found, it was common that no pure rainbow trout were caught. Hybrids were a combination of F1, F2, and commonly backcrosses – that is, a hybrid crossing with a pure parental species. At one site with samples collected at three locations *above* a waterfall, the data showed a decrease in the number of hybrids, as well as a decrease in the degree of hybridization in individual fish, with increased elevation and distance from the barrier.

Population subdivision was substantial, which indicated that among-locality differentiation appears to be concentrated at the level of individual stream or lake, rather than by major drainage system. A more complete discussion of the results can be found in Taylor and Gow (2007, 2009).

Discussion

In their discussion of the results, Taylor and Gow (2007, 2009) suggested that the actual value of any measure of introgression that is used to define genetic “purity” (e.g., 0.99 versus 0.95, etc.) is the subject of some debate (see exchanges by Allendorf et al. 2005; Campton and Kaeding 2005 and the discussion of this in Taylor and Gow 2007). A criterion of 0.99 is the most conservative and is based on the rationale that there is good evidence for natural and historical hybridization between westslope cutthroat trout and rainbow trout (hence a value of 1.0 is not biologically expected), and that such historical effects appear to be at about a level of 0.01 or less.

The results of the genetic analysis clearly indicated that few native pure populations still exist in

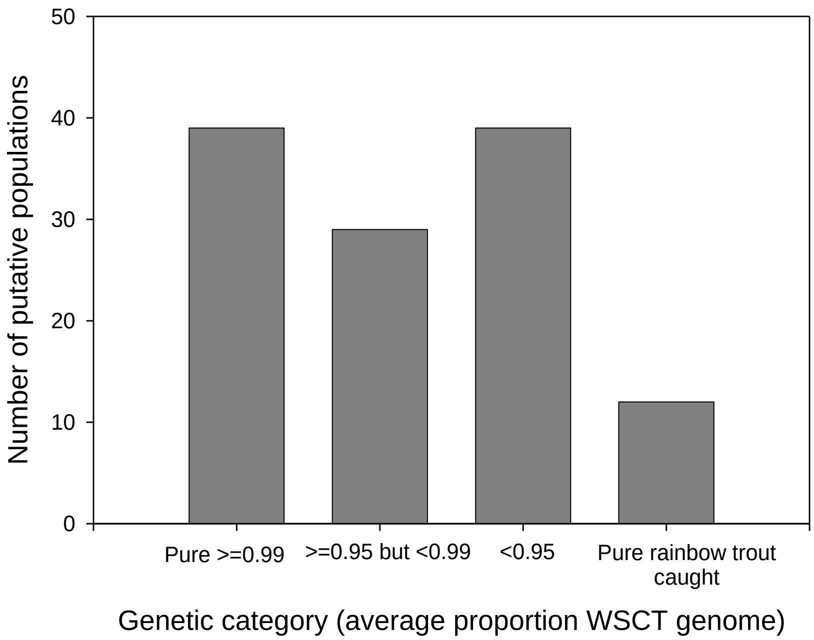


Figure 2. Genetic category of putative populations of westslope cutthroat trout (WSCT) sampled in Alberta, 2006-2009.

provincial waters and those that do are often remnant populations isolated above barriers. This has posed some interesting and complex questions for the recovery team. For example: Is some level of hybridization acceptable? The Committee on the Status of Endangered Wildlife in Canada guidelines suggest that "Populations exhibiting >1% introgression may be considered hybridized and will generally be excluded from COSEWIC status assessments." While the team generally supports adhering to this stringent level of purity, it has also been recognized that less pure populations (e.g., ≥0.95 but <0.99 on average) may be important to species conservation. The Committee on the Status of Endangered Wildlife in Canada guidelines on manipulated populations consider four types of manipulated populations (COSEWIC 2010). The ones of greatest relevance in the case of westslope cutthroat trout in Alberta are (1) introduced/re-introduced and (2) hybrid. In the case of the first type of manipulated population, numerous pure but introduced populations exist in the province. Some of these have established self-sustaining populations, but many are maintained solely by stocking and are provided for recreational purposes. The related guideline for introductions outside their natural range indicates that COSEWIC will generally not include populations resulting from benign outside natural range introductions as part of the wildlife species being assessed, unless there is no suitable habitat remaining within the natural range of the wildlife species in Canada.

In terms of human-mediated hybridization, the guideline suggests that "if introgression is known or suspected, COSEWIC will consider whether it is likely to negatively affect the conservation of the wildlife species." The guidelines state that exceptions may exist where the gene pool of a wildlife species is so small that inbreeding depression is evident and genetic variability cannot be increased using individuals from the same genetic pool.

The issues are somewhat complicated by the fact that the guidelines appear to be primarily directed towards species status assessments and what will or will not be included when applying quantitative criteria to determine status. This may not, however, always correspond with the notions of species conservation, biodiversity and maintaining a diversity of life-history types for a species that currently occupies a small fraction of its historical range.

In the end, the guidelines themselves indicate "some degree of flexibility needs to be retained in the application of the guidelines to allow for the consideration of specific circumstances associated with different species." In addition, advice given from both provincial and federal Species at Risk agencies suggested that a certain amount of flexibility exists for the team in regards to these issues. The team's current direction is to recommend prioritization of native populations based on genetic status. The reality is that for many of these populations, maintenance will become the recovery goal, due to their small size and isolated nature. While the emphasis is on protecting the native pure populations, the team acknowledges the importance of introduced pure populations, as well as those exhibiting slight hybridization.

This direction is reflected in both the draft recovery goal and objectives, as well as the recovery strategy. For example, a recovery objective seeks to determine the role that introduced pure populations may play in the recovery effort. Furthermore, research approaches to address data deficiencies are intended to cover both pure and slightly hybridized populations. Similarly, the team will seek to clarify the distribution and status of introduced populations in and outside of native range. It is felt that this breadth of information is required to obtain a more complete picture of the species status and investigate all opportunities for species maintenance and recovery. The plan is a work in progress and it is the team's goal to have a draft plan produced by the end of 2010 that will prioritize action items for the next 5 years. The plan will provide the strategic framework for operational plans to be delivered on the ground in future years.

Acknowledgements

The author would like to thank Travis Ripley, Jim Stelfox and Shane Petry who provided valuable comments and suggestions on the manuscript.

References

Alberta Sustainable Resource Development and Alberta Conservation Association. 2006. Status of the westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) in Alberta. Alberta Sustainable Resource Development, Wildlife Status Report No. 61, Edmonton.

Alberta Sustainable Resource Development. 2008. Distribution and genetic status of native (not stocked) populations of westslope cutthroat trout sampled between 2000 and 2007. Map prepared by Alberta Fish and Wildlife Division, Calgary, April 2008.

Allendorf, F.W., and R.F. Leary. 1988. Conservation and distribution of genetic variation in a polytypic species, the cutthroat trout. Conservation Biology 2:170-184.

Allendorf, F.W., R.F. Leary, N.P. Hitt, K.L. Knudsen, L. Lundquist, and P. Spruell. 2004. Intercrosses and the U.S. Endangered Species Act: Should hybridized populations be included as westslope cutthroat trout. Conservation Biology 18: 1203-1213.

Allendorf, F.W., R.F. Leary, N.P. Hitt, K.L. Knudsen, M.C. Boyer, and P. Spruell. 2005. Cutthroat trout hybridization and the U.S. Endangered Species Act: one species, two policies. Conservation Biology 19: 1326-1328.

Behnke, R. J. 1992. Native trout of Western North America. American Fisheries Society Monograph 6, Bethesda, MD. xx+275 p.

Behnke, R.J. 2002. Trout and salmon of North America. Simon and Schuster, New York. 359 p.

Bernatchez, L. 1999. Gene diversity analysis of cutthroat trout and rainbow populations from Banff National Park. Département de biologie, Université Laval, Québec, PQ. Report prepared for Banff National Park, Banff, Alberta

Campton, D.E., and L.R. Kaeding. 2005. Westslope Cutthroat Trout, hybridization, and the U.S. Endangered Species Act. Conservation Biology 19: 1323-1325.

Carl, L. M., and J. D. Stelfox. 1989. A meristic, morphometric and electrophoretic analysis of cutthroat trout, *Salmo clarki*, from two mountain lakes in Alberta. Canadian Field-Naturalist 103:80-84.

COSEWIC. 2006a. Database. Committee on the Status of Endangered Wildlife in Canada. URL: http://www.cosewic.gc.ca

COSEWIC. 2006b. COSEWIC assessment and status report on westslope cutthroat trout *Oncorhynchus clarkii lewisi*. Committee on the Status of Endangered Wildlife in Canada. Ottawa, On. Iii+86 p.

COSEWIC. 2010. COSEWIC Guidelines on manipulated populations – Approved April 2010. URL: http://www.cosewic.gc.ca/eng/sct2/sct2_8_e.cfm

Fitch, L. A. 1977-1980. Stream survey reports, Castle and Crowsnest river drainages. Alberta Fish and Wildlife Division, Lethbridge.

Janowicz, M. 2005. Genetic analysis of hybridization between native westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) and introduced rainbow trout (*O. mykiss*) in the eastern slopes of the Rocky Mountains in Alberta. Department of Biology and Environmental Science, Concordia University College of Alberta, Edmonton.

Mayhood, D. W. 1995. The fishes of the Central Canadian Rockies Ecosystem. FWR Freshwater Research Limited Report 950408 prepared for Parks Canada, Banff National Park, P.O. Box 900, Banff, Alberta T0L 0C0. 59 p. Available from: http://www.fwresearch.ca/PDFLibrary.html

Mayhood, D. W., W. Haskins, and M. D. Sawyer. 1997. Cumulative effects on fish. pp. 173-187. *in* Sawyer, M. D., D. W. Mayhood, P. Paquet, R. Thomas, C. Wallis and W. Haskins, editors. Southern East Slopes cumulative effects assessment. Hayduke and Associates Ltd., Calgary Alberta, funded by Morrison Petroleum Ltd, Calgary Alberta. 207p. + appendices.

Mayhood, D. W. 2000. Provisional evaluation of the status of westslope cutthroat trout in Canada. pp. 579-585. *in* Darling, L. M., editor. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, British Columbia, February 15-19, 1999 BC Ministry of Environment, Lands and Parks, Victoria, and University College of the Cariboo, Kamloops, . 2 Volumes. 974 p. Available from: http://www.fwresearch.ca/PDFLibrary.html (Online version includes unpublished maps.)

Mayhood, D.W. 2009. Contributions to a recovery plan for westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) in Alberta. FWR Freshwater Research Limited, Calgary, Alberta. Unpublished report.

McAllister, D. J., F. W. Allendorf and S. R. Phelps. 1981. An analysis of the native and resident cutthroat trout (*Salmo clarki*) in the Bow, Kootenay-Columbia and Waterton river systems. Report prepared by Techman Engineering Ltd. for Parks Canada, Calgary, Alberta. .

Prince, E. E., T. H. McGuire and E. Sisley. 1912. Dominion Alberta and Saskatchewan Fisheries Commission 1910-11. Report and recommendations with appendices. Government Printing Bureau, Ottawa, Ontario.

Radford, D. S. 1975. Oldman River flow regulation: a preliminary study of the fish resources. Alberta Fish and Wildlife Division, Lethbridge.

Radford, D. S. 1977. A report on biological inventories of 17 streams in the Livingstone drainage district. Alberta Fish and Wildlife Division report, Lethbridge.

Robinson, M. D. 2008. Remnant westslope cutthroat trout population inventory for southern Alberta, 2007. Interior Reforestation Co Ltd, Cranbrook, BC, report prepared for Alberta Sustainable Resource Development, Cochrane.

Strobeck, C. 1994. Survey of cutthroat trout in Banff National Park. Department of Biology, University of Alberta. Report prepared for Parks Canada, Banff National Park, Banff.

Taylor, E.B. and J.L. Gow. 2007. An Analysis of hybridization between native westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) and introduced Yellowstone Cutthroat Trout (*O.c. bouvieri*) and Rainbow Trout (*O. mykiss*) in Canada's Mountain Parks and adjacent watersheds in Alberta. Final report to Parks Canada and Alberta Fish and Wildlife. Department of Zoology, Biodiversity Research Centre and Native Fishes Research Group, University of British Columbia. Vancouver.

Taylor, E.B. and J.L. Gow. 2009. An Analysis of hybridization between native westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) and introduced Yellowstone Cutthroat Trout (*O.c. bouvieri*) and Rainbow Trout (*O. mykiss*) in Canada's Mountain Parks and adjacent watersheds in Alberta: Summer 2007 Data. Addendum to 2007 Report to Parks Canada and Alberta Fish and Wildlife. Department of Zoology, Biodiversity Research Centre and Native Fishes Research Group, University of British Columbia. Vancouver

.

CUTTHROAT TROUT TAXONOMY: EXPLORING THE HERITAGE OF COLORADO'S STATE FISH

Kevin B. Rogers

Aquatic Research Group, Colorado Division of Wildlife, 925 Weiss Dr., Steamboat Springs, Colorado 80477, USA, kevin.rogers@state.co.us

ABSTRACT — Prior to recent molecular testing, larger spots and more scales above the lateral line were phenotypic traits associated with greenback cutthroat trout *Oncorhynchus clarkii stomias* as compared to Colorado River cutthroat trout *O. c. pleuriticus*. However, these two subspecies could not be separated consistently on the basis of those characteristics. As a result, geographic range had become the default approach for establishing subspecies designation. In 2007, researchers at the University of Colorado and others used mitochondrial DNA (mtDNA), microsatellites, and amplified fragment length polymorphisms (AFLPs) to suggest there was a genetic basis for separating greenback cutthroat trout from Colorado River cutthroat trout. The primary concern raised by that study was that five of the nine greenback cutthroat trout populations they examined actually displayed genetic fingerprints more similar to Colorado River cutthroat trout from Trappers Lake, Colorado (Lineage CR). A finding that attracted less attention was the discovery of an alleged greenback cutthroat trout population west of the Continental Divide near Gunnison, Colorado, in West Antelope Creek. Recovery Team partners used the same AFLP test to canvass cutthroat trout populations for genetic purity across Colorado. Results indicate that the West Antelope Creek population is not unique, as 46 additional populations sharing the "greenback" genetic fingerprint (Lineage GB) have now been identified west of the Continental Divide. It is therefore questionable whether West Antelope Creek fish are really greenback cutthroat trout as suggested in the 2007 study, or whether they simply represent diversity within Colorado River cutthroat trout. Given their broad geographic distribution and lack of a robust East Slope source from which to stock, it seems unlikely that these Lineage GB populations were established by anthropogenic means. A more parsimonious explanation would suggest they are aboriginal to Colorado's West Slope. Interestingly, no Lineage GB populations have been discovered in the White and Yampa river basins. This region appears to be a stronghold for Lineage CR cutthroat trout and coincides with the native range of Colorado's only other native salmonid, the mountain whitefish *Prosopium williamsoni*. Perhaps Lineage GB fish were already present in southwestern Colorado when Lineage CR and mountain whitefish invaded the state from the Green River drainage.

INTRODUCTION

There has been a long-standing interest in the taxonomy of cutthroat trout in Colorado ever since the rediscovery and listing of greenback cutthroat trout *Oncorhynchus clarkii stomias* as endangered under the Endangered Species Act in 1973 and subsequent down-listing to threatened status in 1978. No fewer than 72 reports have been published since then discussing genetic purity, introgression, and relatedness. Much of this work has been summarized in several books (Gresswell 1988; Behnke 1992; Behnke 2002). Phenotypic traits typically associated with greenback cutthroat trout are larger spot size and more scales above the lateral line and in the lateral series as compared to Colorado River cutthroat trout *O. c. pleuriticus* (Behnke 1992). However, these two subspecies cannot be separated definitively on the basis of these characteristics (Behnke 1992; Behnke 2002). Further, Behnke maintains that a modern taxonomist would not separate the greenback cutthroat trout from Colorado River cutthroat trout (R. J. Behnke, personal communication). Given the difficulty separating the two subspecies using visual characteristics, geographic range became the default criterion for establishing subspecies designation.

Though early molecular work also failed to reliably distinguish these two subspecies (Behnke 2002), more recent studies of molecular data, including mitochondrial DNA (mtDNA), microsatellites, and amplified fragment length polymorphisms (AFLPs) suggested that indeed there was a genetic basis for separating greenback cutthroat trout from Colorado River cutthroat trout (Metcalf et al. 2007). An important finding of the Metcalf et al. study was that many of the putatively pure greenback cutthroat trout populations east of the Continental Divide were genetically more similar to Colorado River cutthroat trout from Trappers Lake, Colorado (Lineage CR), than to other greenback cutthroat trout populations such as those found in Severy Creek, Colorado (Lineage GB). This was particularly troubling since mechanisms were in place to deliver pure Lineage CR to the East Slope. From 1903 through 1938 what could have been well in excess of 40 million pure Colorado River cutthroat trout were produced at Trappers Lake, Colorado (Rogers 2008). Millions more were produced on the south slope of Pikes Peak (Rogers and Kennedy 2008). The fate of many of these fish remains a mystery. The long-standing notion that fishless waters above barriers to immigration were likely stocked from nearby waters may not hold, as fingerlings produced at Trappers Lake and other state and federal spawn operations were readily available, and obviated the need for capturing wild fish. It is certainly conceivable that some pure cutthroat trout populations east of the Continental Divide were founded in historically fishless waters from progeny of distant spawn operations.

The publication of the Metcalf et al. (2007) paper was followed by an international media firestorm critical of recovery program efforts for allegedly using the “wrong” fish for founding new populations of cutthroat trout in greenback cutthroat trout recovery waters. The Greenback Recovery Team has always used the best available science to guide recovery efforts, recognizing that science is a dynamic field. As such, the most prudent management direction can and often does change over time. Although the multi-agency Greenback Recovery Team had initiated development of a long-range greenback cutthroat trout management plan that would allow delisting after decades of progress, unresolved taxonomic issues regarding what constitutes a greenback cutthroat trout ground recovery efforts to a halt.

A finding of Metcalf et al. (2007) that attracted less attention was the discovery of an alleged greenback cutthroat trout population west of the Continental Divide near Gunnison, Colorado, in West Antelope Creek. Early results from concurrent studies conducted by the Colorado Division of Wildlife through a private genetics lab using AFLP markers indicated that in fact the West Antelope Creek population was not unique. By 2008, a dozen additional populations displaying the Lineage GB fingerprint were identified west of the Continental Divide (Rogers 2008). Those findings led the Recovery Team to question whether the West Antelope Creek fish were really greenback cutthroat trout as suggested by Metcalf et al. (2007), or whether they represented a diverse lineage endemic to the western slope of Colorado. Interest in resolving that dilemma lead to the major molecular survey efforts by Greenback Recovery Team and Colorado River Cutthroat Trout Conservation Team members discussed here.

Methods

Tissue Collection

Fin tissues from over 8,200 fish were collected from 366 populations of cutthroat trout in Colorado and southern Wyoming. A small piece (1 cm^2) fin was collected from the top of the caudal fin from each fish, as this fin regenerates rapidly, is assumed to be less critical for digging redds, and provides adequate tissue volume even on small fish (Rogers 2007). Fins were stored individually in 15-mL conical centrifuge tubes filled with 80% reagent grade ethanol until processing. In addition, UTM coordinates, photographs, and fish lengths were recorded.

DNA Isolation and Evaluation

Tissue samples were delivered to Pisces Molecular (Boulder, Colorado) for DNA isolation and testing with the same AFLP procedure describe in Metcalf et al. (2007). Cutthroat trout DNA was extracted from fin clips using a proteinase K tissue lysis and spin-column DNA purification protocol (DNeasy Tissue Kit, Qiagen, Inc. Chatsworth, CA).

The DNA was then amplified using a polymerase chain reaction (PCR) to produce AFLP marker fragments. These fragments were separated and sized on an ABI 3130 sequencer (Applied Biosystems, Foster City, California).

Using Genemapper 4.0 (Applied Biosystems), a genetic fingerprint was produced for each individual sample by scoring for the presence or absence of a standardized set of 119 markers between 50 and 450 base pairs in size generated from a reference set of cutthroat trout populations (Rogers 2008). The genetic fingerprints of individuals in the test population were compared to those found in the reference populations using a Bayesian approach for identifying population clusters (Pritchard et al. 2000). The program STRUCTURE 2.2 (Falush et al. 2007; Pritchard et al. 2007) was used to evaluate similarity between the test individuals and the reference populations. Reference populations were selected and grouped by their mtDNA lineage (Metcalf et al. 2007), and not necessarily by geographic or historic subspecies classifications. The similarity or dissimilarity was scored as the admixture proportion, or the probability that each test individual shares a genetic background with each of the cutthroat trout subspecies reference population groups. These proportions are expressed as q values for each subspecies. These q values were obtained by running STRUCTURE ten times for each population of interest using a burn-in of 50,000 steps followed by 50,000 Monte Carlo Markov Chain replicates. Average q values from the run with the highest log likelihood (Pritchard and Cowley 2007) were used to generate the admixture proportions for the population in question.

Populations showing greater than 20% admixture (where q values were less than 0.80) with Yellowstone cutthroat trout *O. c. bouvieri* , rainbow trout *O. mykiss* or the two lineages native to Colorado (Lineage CR and Lineage GB) were excluded from further consideration. Locations of the populations were plotted to discern spatial relationships. The number of populations represented by each lineage was calculated for each major watershed-based geographic management unit (GMU) across the range of Colorado River cutthroat trout (Hirsch et al. 2006).

RESULTS

Extensive molecular surveys of the 366 populations of cutthroat trout in Colorado and the Little Snake River drainage in southern Wyoming have identified 156 streams containing cutthroat trout west of the Continental Divide that displayed q values higher than 0.80 as measured with AFLP markers (Rogers 2008). This approach identified 37 populations as Lineage GB. When combined with an additional 9 populations identified from molecular work conducted at Brigham Young University (D. Shiozawa, Brigham Young University, personal communication) and one population at the University of Colorado (Martin 2008), it appears that at least 47 populations of Lineage GB exist west of the Continental Divide at the time of this writing. These populations are distributed across 14 counties in western Colorado and one in San Juan County, Utah (Figure 1). Populations are concentrated around the Grand Mesa (Figure 1) but range into the headwaters of the Colorado, Gunnison, and Dolores drainages as well. No Lineage GB populations were found in the Yampa, White, or Little Snake river basins where Lineage CR is pervasive (Figure 2). In fact, 59 of the 109 Lineage CR populations identified in this survey were located in the Yampa GMU, though good numbers (36) were also found in the Upper Colorado GMU (Figure 3).

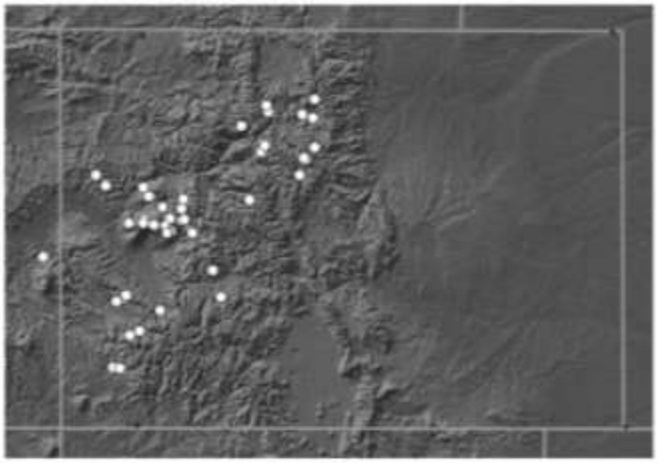


Figure 1: Geographic location of 47 Lineage GB cutthroat trout populations identified west of the Continental Divide as of August 2010.

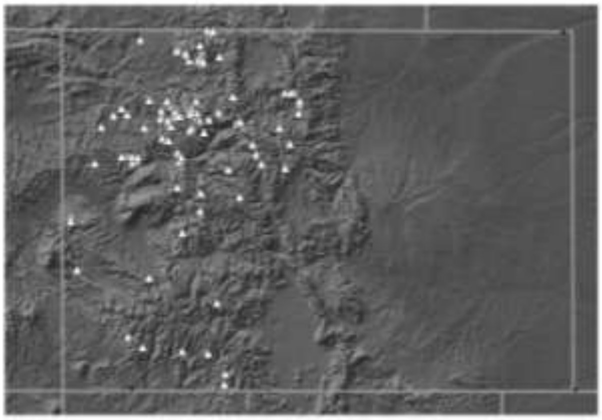


Figure 2: Geographic location of 109 Lineage CR cutthroat trout populations identified west of the Continental Divide as of August 2010.

Discussion

Given their broad geographic distribution and lack of a robust East Slope wild spawn operation from which to gather fertilized eggs, it seems unlikely that Lineage GB populations were established west of the Continental Divide by anthropogenic means as suggested by Metcalf et al. (2007). A more parsimonious explanation is that these fish are aboriginal at least to the Gunnison and Dolores basins, establishing east of the Continental Divide either on their own or with the aid of early fish culturists. As such, Lineage GB may represent a divergent lineage reflecting a complex past within Colorado River cutthroat trout. This scenario is consistent with Evans and Shiozawa's (2000) assertion that Colorado River cutthroat trout were

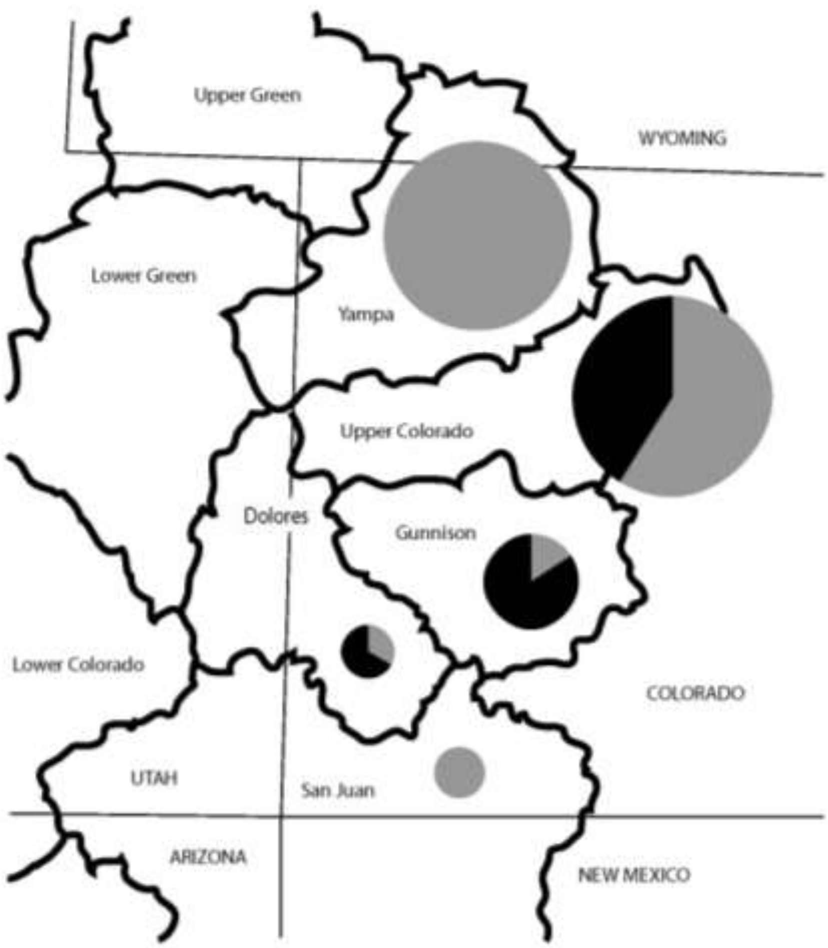


Figure 3: The area of each pie is proportional to the number of cutthroat trout populations from each GMU that displayed less than 20% admixture with rainbow trout or Yellowstone cutthroat trout. Wedges from each pie represent the number that were Lineage CR (Gray) or Lineage GB (black). This study did not cover the Upper Green, Lower Green, or Lower Colorado GMUs.

established first by an ancient invasion from the west which was then masked in many populations by subsequent invasions from the north out of Wyoming down the Green River drainage.

Of particular interest is the lack of Lineage GB populations in the Yampa GMU (White, Yampa, and Little Snake river drainages in northwestern Colorado and southern Wyoming). This area appears to be a stronghold for Lineage CR cutthroat trout and coincides with the native range of Colorado's only other native salmonid – the mountain whitefish *Prosopium williamsoni*. Whiteley et al. (2006) conducted a range-wide survey of mountain whitefish and found five genetically divergent assemblages that were organized around major drainages. Mountain whitefish in Colorado are likely most closely related to the Snake River assemblage perhaps invading from the north down the Green River drainage along with Lineage CR following the recession of the last ice age. It is noteworthy that mountain whitefish have become very common in the Colorado and Roaring Fork rivers following introductions in the 1940s, even though they are not native to those drainages (Feast 1938). Clearly, the habitat is not limiting, so the absence of mountain whitefish historically suggests that something else excluded them. Perhaps the desert confluence of the Colorado and Green Rivers provided a thermal barrier to movement out of the Green River into the Colorado River. Interestingly, mountain suckers *Catostomus platyrhynchus* are similarly restricted to the Yampa GMU (Woodling 1985), perhaps reflecting the invasion of an entire assemblage of fish species at the same time.

Whether Lineage GB, Lineage CR, or both are native to the Upper Colorado GMU is still difficult to discern, as both lineages are present in large numbers (Figure 3). Many of the Lineage GB populations are found in remote locations that might not have attracted much attention from early fish managers. Much of the Upper Colorado River is prime trout habitat, and it was those prime habitats that apparently attracted the subsequent stocking of potentially hybridizing species leaving remnant populations of pure cutthroat trout spread around the periphery of the range (Hirsch et al. 2006). While that might argue for Lineage GB as the aboriginal fish to the Upper Colorado GMU, it is true that unlike mountain whitefish, cutthroat trout are capable of invading the uppermost tributaries of headwater streams. That ability makes them much better candidates for successful headwater transfers. Certainly there are numerous shallow divides near the headwaters of the White and Yampa rivers that could have acted as conduits for the natural migration of Lineage CR into the Colorado River Basin.

The presence of only Lineage CR fish in the San Juan River drainage was curious since that part of the state lies farthest from the Lineage CR stronghold in the Yampa GMU, and close to the Dolores and Gunnison GMUs where Lineage GB is pervasive (Figure 3). Metcalf et al. (2007) already suggested that one of these populations (East Fork Piedra River) was likely founded by stocking given the similarity of its genetic fingerprint to geographically remote populations to the north. Further research will be necessary to determine if the seven other populations in the San Juan GMU identified in this report also share DNA similar to more northerly populations such as Trappers Lake that provided large numbers of pure cutthroat trout to the state hatchery system, or whether they harbor unique haplotypes, suggesting remnant genetic diversity across the landscape.

Given the illustrious stocking history in Colorado over the last century (Wiltzius 1985), it will be challenging to completely resolve the native distribution of these lineages. Ongoing research at the University of Colorado studying museum specimens of cutthroat trout collected in the late 1800s will hopefully shed some light on the historic distributions of these fish prior to the bulk of fish culture activity in the state. A Colorado State University study exploring different visual characteristics between the lineages will also be illuminating. Piecing together the taxonomic past of these fish will be critical for providing information that will allow recovery efforts to resume.

Acknowledgements

Dedicated biologists from state and federal agencies are thanked for securing tissue samples analyzed in the document. The Colorado Wildlife Commission, Sherman Hebein, Mark Jones, Mike Japhet, Greg Gerlich, and John Alves (Colorado Division of Wildlife); Christine Hirsch, Rick Henderson, Greg Eaglin, Chris James, and Clay Speas (U. S. Forest Service); Tom Fresques, and Patrick Lionberger (Bureau of Land Mangement); and David Nickum (Trout Unlimited) are all thanked for their financial contributions toward processing of the tissues covered in this report. I also wish to thank John Wood, Janet Epp, and their diligent team at Pisces Molecular for conducting the molecular analyses used in this study and for providing results in a timely fashion. This manuscript was greatly improved with helpful reviews by Andrew Martin, Jay Thompson, Mark Jones, and Bruce Rosenlund.

Literature Cited

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society Monograph 6.

Behnke, R. J. 2002. Trout and salmon of North America. The Free Press.

Evans, R. P., and D. K. Shiozawa. 2000. The genetic status of selected cutthroat trout populations in Colorado. May 1, 2000. Brigham Young University, Provo, Utah.

Falush, D., M. Stephens, and J. K. Pritchard. 2007. Inference of population structure using multilocus genotype data: dominant markers and null alleles. Molecular Ecology Notes.

Feast, C. N. 1938. A preliminary study of the proposed whitefish transplanting in western Colorado waters. United States Forest Service report. Available: http://wildlife.state.co.us/Research/Aquatic/MountainWhitefish/ (April 2009)

Gresswell, R. E. 1988. Status and management of interior stocks of cutthroat trout. American Fisheries Society Symposium 4, Bethesda, Maryland.

Hirsch, C. L., S. E. Albeke, and T. P. Nesler. 2006. Range-wide status of Colorado River cutthroat trout (*Oncorhynchus clarkii pleuriticus*): 2005. Colorado Division of Wildlife, Denver. Available: http://wildlife.state.co.us/research/aquatic/cutthroattrout (April 2008).

Martin, A. 2008. Genetic identification of cutthroat trout in Rocky Mountain National Park based on mtDNA and microsatellite analysis. University of Colorado, Boulder.

Metcalf, J. L., V. L. Pritchard, S. M. Silvestri, J. B. Jenkins, J. S. Wood, D. E. Cowley, R. P. Evans, D. K. Shiozawa, and A. P. Martin. 2007. Across the great divide: genetic forensics reveals misidentification of endangered cutthroat trout populations. Molecular Ecology 16:4445-4454.

Pritchard, J. K., M. Stephens, and P. Donnelly. 2000. Inference of population structure from multilocus genotype data. Genetics 155:945-959.

Pritchard, J. K., X. Wen, and D. Falush. 2007. Documentation for structure software: Version 2.2. University of Chicago. Available: http://pritch.bsd.uchicago.edu/software. (June 2007).

Pritchard, V. L., K. Jones, and D. E. Cowley. 2007. Estimation of introgression in cutthroat trout populations using microsatellites. Conservation Genetics 8:1311-1329.

Rogers, K. B. 2007. A suggested protocol for collecting cutthroat trout tissues for subsequent genetic analysis. Colorado Division of Wildlife, Fort Collins.

Rogers, K. B. 2008. Using amplified fragment length polymorphisms to characterize purity of cutthroat trout in Colorado: results from 2007. Colorado Division of Wildlife, Fort Collins.

Rogers, K. B., and C. M. Kennedy. 2008. Seven Lakes and the Pike's Peak native (PPN): history and current disposition of a critical cutthroat trout brood stock. Colorado Division of Wildlife, Fort Collins.

Whiteley, A. R., P. Spruell, and F. W. Allendorf. 2006. Can common species provide valuable information for conservation? Molecular Ecology 15:2767-2786.

Wiltzius, W. J. 1985. Fish culture and stocking in Colorado, 1872-1978. Colorado Division of Wildlife, Division Report 12, Fort Collins, Colorado

Woodling, J. 1985. Colorado's little fish: a guide to the minnows and other lesser known fishes in the state of Colorado. Colorado Division of Wildlife, Denver.

Cutthroat Trout Phylogenetic Relationships with an Assessment of Associations Among Several Subspecies.

Dennis K Shiozawa[1,3], R. Paul Evans[3], Peter Unmack[2], Andrew Johnson[2], and Jason Mathis[2]

[1,2]Curator, Ichthyology Collection, Monte L. Bean Life Science Museum, Brigham Young University, Provo, Utah, 84602, shiozawa@byu.edu

[2] Department of Biology, Brigham Young University, Provo, Utah, 84602

[3] Department of Microbiology and Molecular Biology, Brigham Young University, Provo, Utah, 84602, evansp@byu.edu

ABSTRACT — We use multiple protein coding mitochondrial genes to investigate broader phylogenetic patterns across the range of *Oncorhynchus clarkii*. Three moderate to strongly resolved main lineages are identified. One forms a basal cluster consisting of coastal cutthroat trout *O. c. clarkii*, Lahontan cutthroat trout *O. c. henshawi*, and westslope cutthroat trout *O. c. lewisi*. The remaining two lineages are in a well supported clade. One of these includes the Yellowstone cutthroat trout *O. c. bouvieri* and Bonneville cutthroat trout *O. c. utah* of the Bear River Basin of Utah and Idaho. The other lineage consists of Colorado River *O. c. pleuriticus*, greenback *O. c. stomias*, Rio Grande *O. c. virginalis*, and Bonneville (main Bonneville Basin) cutthroat trout. The subspecies are strongly resolved, but separation of subspecies within lineages is not strongly supported by bootstrap values. Two groups, the Yellowstone and Bear River Bonneville cutthroat trout, and the greenback cutthroat trout have proven to be problematic relative to their taxonomic status as well as original distribution. These are further examined to help resolve their biogeographical history.

Introduction

The phylogeography of aquatic organisms in western North America has been strongly influenced by the complex geological history of the region (Hubbs and Miller 1948; Smith 1981; Taylor 1985). Different dispersal capabilities of taxa, both invertebrate and vertebrate, appear to be important in the dispersal paths that the organisms have taken (Kauwe et al. 2004; Houston et al 2010; Stutz 2010). Fishes, because they require sufficient water to avoid seasonal and long-term temperature extremes and desication, carry genetic signals that are more constrained over geological history than are those of aquatic invertebrates (Taylor 1985; Stutz 2010).

Our work on native fishes in the basins of Western North America has also documented further subdivisions of dispersal ability among fish taxa. Cyprinids tend to show pre-Pleistocene dispersal patterns, in the range of 5 to 15 million years (Billman et al 2010; Houston et al 2010). Cottids show both recent and pre-Pleistocene dispersal as do catostomids (Shiozawa, Unmack, and Evans unpublished data). Cutthroat trout, *Oncorhynchus clarkii*, have generally been assumed to have dispersed through the west much more recently (Hubbs and Miller 1948; Behnke 1992, 2002; Wilson and Turner 2009). Their ability to leap barriers and to inhabit cold, low order, high elevation streams increases their likelihood of dispersing through headwater capture events that tend to exclude other fishes. Yet that does not prevent trout from also dispersing with other fish species; hence, a general phylogeny of cutthroat trout may give insight into both recent transfer events as well as less frequent events that allow major faunal exchanges between basins.

Beyond their role in recording multiple windows of geologic history, cutthroat trout are important relative to current conservation issues. Several subspecies are threatened, and the remainder is classified as species of concern by state and federal agencies. A better understanding of their phylogenetic relationships, between and within subspecies, will improve the precision of management decisions. We need to understand exactly what comprises the lineages that are being managed and how much genetic variation exists within those lineages. Improper or incomplete identification and classification can lead to critical populations being either omitted from consideration or lumped together with less critical populations, thus generating a false sense of stability or robustness of populations or subspecies.

Attempts to understand the phylogentic relationships among cutthroat trout have been made with a variety of approaches. Behnke (1992, 2002) compiled a cutthroat trout phylogeny based upon a combination of morphological characters, meristics, karyotypes, and geographic proximity. These relationships have been further examined by molecular approaches, including allozymes (Loudenslager and Gall 1980; Allendorf and Leary 1988) and DNA-based phylogenies (Smith et al 2002; Pritchard et al 2009; Wilson and Turner 2009). It appears that the associations among cutthroat trout subspecies, whose phylogenies are shallow in time, are resolvable with mitochondrial DNA (mtDNA).

Our goals are (1) to generate a comprehensive phylogeny within the cutthroat trout, and (2) to further examine relationships among several problematic subspecies using haplotype networks based on multiple mtDNA genes.

Methods

DNA isolation and sequencing

We selected four genes, NADH dehydrogenase subunit 1 (ND1), NADH dehydrogenase subunit 2 (ND2), ATPase, and cytochrome b (Cytb), for amplification. Tissues and DNA archived in the Monte L. Bean Life Science Museum fish tissue collection at Brigham Young University (BYU) were used for the samples presented in this paper. We extracted total DNA from muscle or fin tissue using the DNeasy Tissue Kit (QIAGEN Inc., Chatsworth, CA). Nuclear and mitochondrial gene regions were amplified by polymerase chain reaction using appropriate primer pairs. Final concentrations for polymerase chain reaction (PCR) components per 25 µL reaction were as follows: 25 ng template DNA, 0.25 µM of each primer, 0.625 units of Taq DNA polymerase, 0.1 mM of each dNTP, 2.5 µL of 10X reaction buffer and 2.5mM MgCl2. Amplification parameters were as follows: 94°C for 3 min followed by 35 cycles of 94°C for 30 s, 48°C for 30 s, and 72°C for 60 s, and 72°C for 7 min. We examined PCR products on a 1% agarose gel using SYBR safe DNA gel stain (Invitrogen, Eugene, OR). Nuclear genes were re-amplified using nested PCR using various primer combinations. We purified PCR products using a Montage PCR 96 plate (Millipore, Billerica, MA). Sequences were obtained via cycle sequencing with Big Dye 3.0 dye terminator ready reaction kits using 1/16th reaction size (Applied Biosystems, Foster City, CA). Sequencing reactions were run with an annealing temperature of 52°C following the ABI manufacturer's protocol. We purified sequenced products using sephadex columns. Sequences were obtained using an Applied Biosystems 3730 XL automated sequencer at the Brigham Young University DNA Sequencing Center.

Analysis of DNA sequence data

DNA sequences were edited using Chromas Lite 2.0 (Technelysium, Tewantin, Queensland, Australia) or Sequencher 4.0 (Gene Codes Corporation, Michigan, United States) and imported into BioEdit 7.0.5.2 (Hall 1999) then aligned by eye. Protein coding sequences were checked for unexpected frame shift errors or stop codons in Mega 4.0 (Tamura et al. 2007) or Sequencher 4.0. For ML analysis we used RAxML 7.2.3 (Stamatakis 2006a; Stamatakis et al.2008) by bootstrapping with 1000 replicates using GTRGAMMA model (Stamatakis 2006b) on the CIPRES cluster at the San Diego Supercomputer Center to place the new sequences into context with our existing sequences from a range of western USA salmonids. Haplotype networks were also generated using the program TCS v1.21 (Templeton et al. 1992; Clement et al. 2000) for greenback cutthroat trout, *Oncorhynchus clarkii stomias*, haplotypes.

Results

The combined mitochondrial genes, ND1, ND2, ATPase, and Cytb found 193 phylogenetically informative, 237 uninformative, and 3,219 invariant nucleotide positions. A maximum likelihood phylogeny (Figure 1) based on 3,649 nucleotide sequences generated three supported clades. One includes coastal cutthroat trout *O. c. clarkii* westslope cutthroat trout *O. c. lewisi* and Lahontan cutthroat trout *O. c. henshawi* supported with a ML bootstrap value of 62%. The remaining extant interior cutthroat trout separate into two well supported clades, a Yellowstone cutthroat trout *O. c. bouvieri*, and Bear River

Bonneville cutthroat trout *O. c. utah* clade (ML support 100%), and a Colorado River cutthroat trout *O. c. pleuriticus* greenback cutthroat trout, Bonneville cutthroat trout, and Rio Grande cutthroat trout *O. c. virginalis* clade.

The TCS haplotype network combining the Bear River cutthroat trout and the Yellowstone cutthroat trout separates them into two distinct groups (Figure 2). The Yellowstone cutthroat trout from both the Yellowstone River Basin and the Snake River form a closely related network, sharing some haplotypes in both basins. The Bear River Bonneville cutthroat trout forms a separate network 9 to 12 steps removed from the nearest Yellowstone cutthroat trout haplotype. Three haplotypes from the Snake River are aligned peripherally with the Bear River clade. These three haplotypes appear to be derived and are four to five steps removed from the nearest Bear River haplotypes.

Sufficient information existed to generate an informative haplotype network for the greenback cutthroat trout (Figure 3). Both ND4L/4 and ND2 contain the majority of the parsimony informative sites (those providing information about phylogenetic relationships) while the other genes only provide a single parsimony informative base change each. Combining ND4L/4, ND2, ATPase, and Cytb resulted in seven haplotypes. These were used for generating the data for the haplotype network. The greenback cutthroat trout TCS network showed haplotypes separated by from one to15 steps. The most similar haplotypes were from North Taylor and Como creeks in the Arkansas River Basin, where a single nucleotide difference of the 3,894 nucleotides separates the two. These two haplotypes form the most distant group, Como Creek being 10 steps away from the next most similar haplotype, Cabin Creek (Figure 3). The remaining haplotypes range from nine to four steps away from one another. Most of these are from the Colorado River Basin but one, the South Prong of Hayden Creek, is an Arkansas River Basin population. It is a minimum of seven steps away from the Colorado River haplotypes.

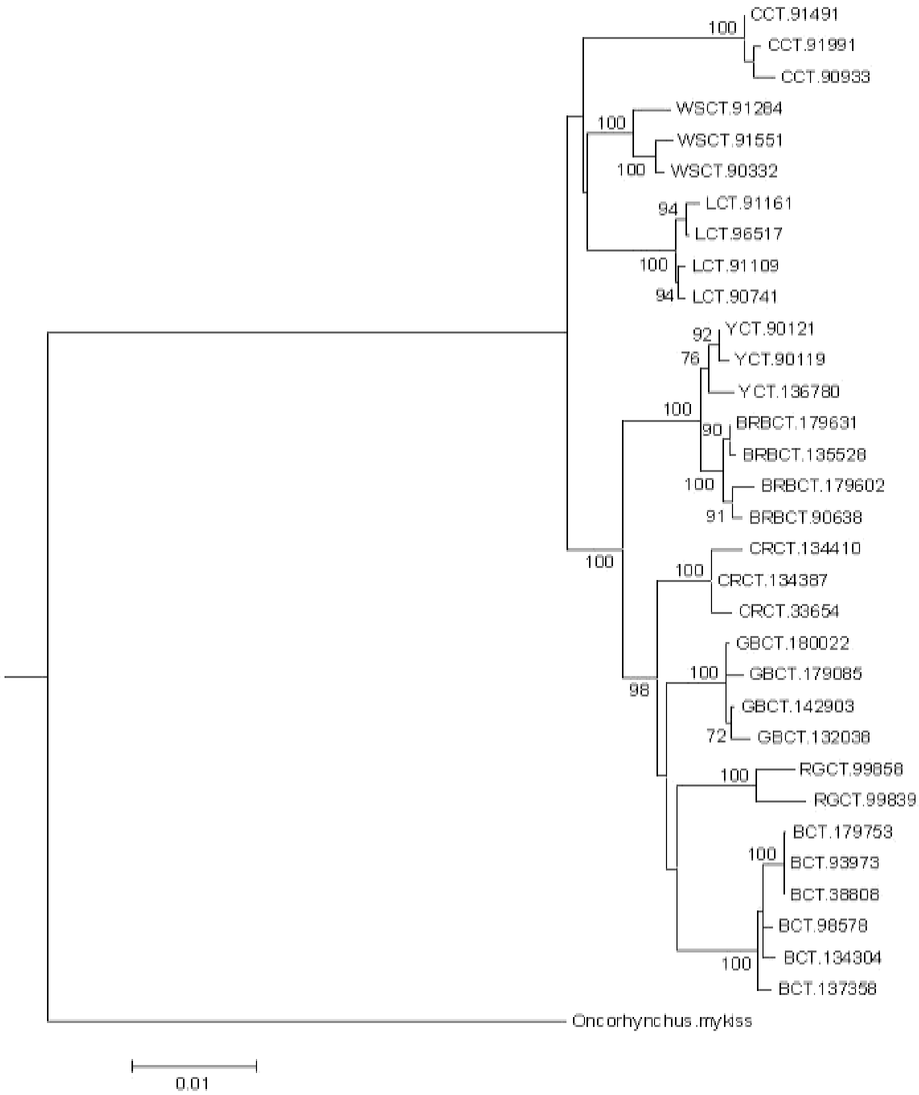


Figure 1. Maximum Likelihood *O. clarkii* phylogeny based on 3,649 base pairs of the mitochondrial genome. Cutthroat trout subspecies abbreviations: BCT (Bonneville cutthroat trout), RGCT (Rio Grande cutthroat trout), GBCT (greenback cutthroat trout), CRCT (Colorado River cutthroat trout) YCT (Yellowstone cutthroat trout), BRBCT (Bear River Bonneville cutthroat trout), LSRCT (Lower Snake River Plain cutthroat trout), CCT (coastal cutthroat trout), WSCT (westslope cutthroat trout), LCT (Lahontan cutthroat trout). Numbers are nodal support (in %), nodal support less than 50% not shown.

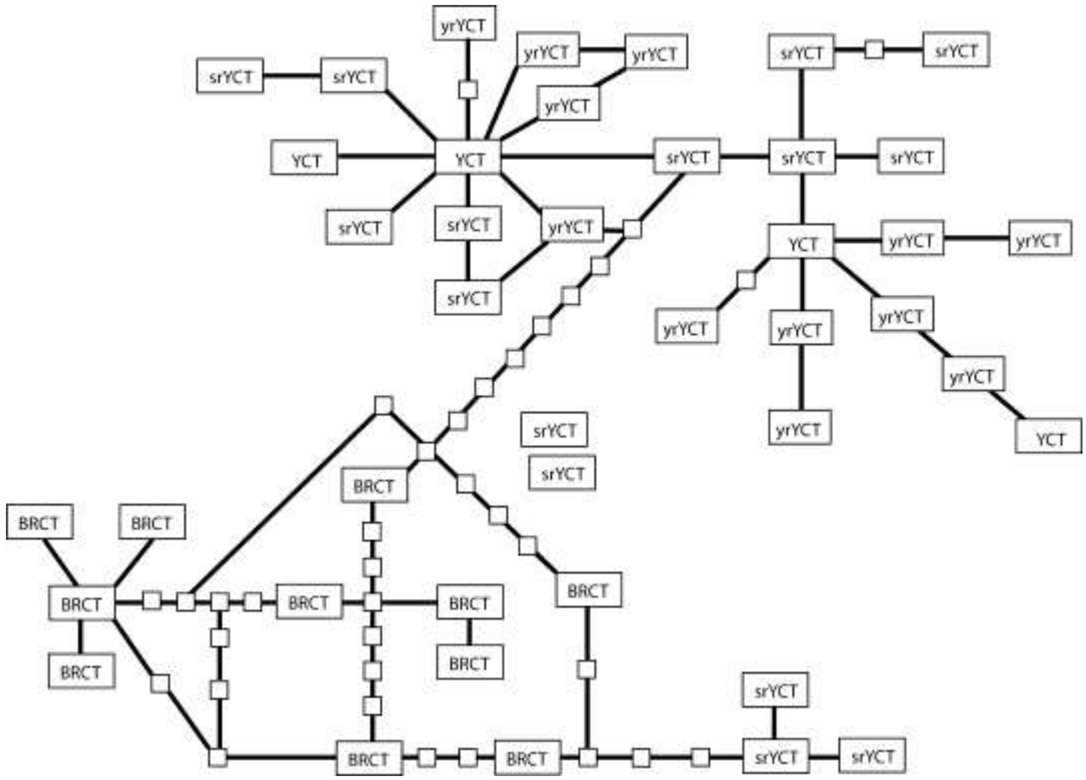


Figure 2. TCS network from 2400 base pairs of ND1 and ND2 with 98% connection confidence limits for Yellowstone cutthroat trout and Bear River Bonneville cutthroat trout haplotypes. BRCT = Bear River form of the Bonneville cutthroat trout from the Bear River Drainage Basin of the Bonneville Basin; YCT = Yellowstone cutthroat trout in both the Yellowstone River Basin and the Snake River Basin; srYCT = Yellowstone cutthroat trout only in the Snake River Basin; yrYCT = Yellowstone cutthroat trout only in the Yellowstone River Basin. Small squares are inferred transitional haplotypes.

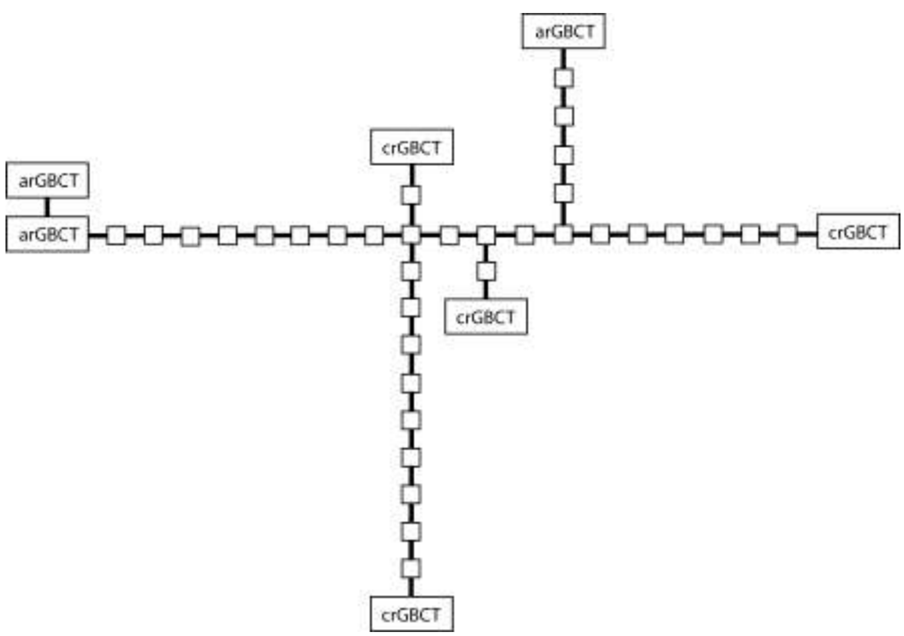


Figure 3. TCS network of greenback cutthroat trout haplotypes. crGBCT = Colorado River Basin greenback cutthroat trout; arGBCR = Arkansas River Basin greenback cutthroat trout. Based on ND2, ATPase, cytB, and last half of ND4 (3,894 bp), run with 98% connection confidence. Small squares are inferred transitional haplotypes.

Discussion

Phylogenetic Relationships

The general phylogeny (Figure 1) indicates that the early establishment of cutthroat trout in western North America focused on two groups, a coastal-inland complex ancestral to the coastal cutthroat trout, Lahontan cutthroat trout, and westslope cutthroat trout, and an interior complex, which gives rise to the Yellowstone cutthroat trout, Bonneville cutthroat trout, greenback cutthroat trout, Colorado River cutthroat trout, and Rio Grande cutthroat trout. The weak nodal support for separating the coastal-Lahontan-westslope lineages is likely due to a rapid diversification of these groups over a relatively short time interval, which may imply that the basal group among these three is not resolvable.

The remaining complex split into two lineages, one which would eventually give rise to the Yellowstone cutthroat trout and the other which would generate the remaining interior cutthroat trout. The latter lineage's relationships weakly supports the Colorado River cutthroat trout

as the basal subspecies, but the associations among the other subspecies are not well resolved with this analysis, even though each subspecies forms a strongly supported clade. Because the Bonneville and Colorado River basins were both adjacent to the early Snake River Basin, cutthroat trout may have dispersed into these basins almost simultaneously. Trout dispersal that generated the Rio Grande cutthroat trout and greenback cutthroat trout may have come from either the Colorado River or Bonneville basins. Our data are equivocal on this point.

The Bear River form of the Bonneville Cutthroat Trout

The Bonneville cutthroat trout is classically defined as the native trout in the Bonneville Basin. It has been hypothesized to have originated from an invasion into the Bonneville Basin with the late Pleistocene capture of the Bear River from the Snake River drainage (Hubbs and Miller 1949; Behnke 1992, 2002). Yet Loudenslager and Gall (1980) and Martin et al. (1985), using allozymes, showed that the Bonneville cutthroat trout could be separated into two lines, one with close associations to the Yellowstone cutthroat trout, as would be expected from a relatively recent transfer from the Snake River system, and the other more closely related to the Colorado River cutthroat trout. The Bonneville cutthroat trout lineage most closely related to the Yellowstone cutthroat trout occurs in the Bear River system. The other lineage is found throughout the main Bonneville Basin. This indicates that the Bonneville cutthroat trout in the Bear River is a remnant of the cutthroat trout inhabiting the Bear River Basin while it was still part of the upper Snake River drainage. It is genetically distinct from the main Bonneville Basin cutthroat trout.

The TCS network, based on 2,500 bp of the ND1&2 genes of the mtDNA genome, demonstrates a distinct separation of the Bear River Basin Bonneville cutthroat trout (BRCT) from the Yellowstone cutthroat trout in the Snake River (srYCT) and Yellowstone River (yrYCT) basins. The relationships suggest that a clade focused in the Bear River was differentiated from the trout in the upper Snake River prior to the transfer of the Bear River into the Bonneville Basin. Some limited gene flow took place, with cutthroat trout from the Bear River lineage either dispersing into the Snake River or alternatively the Bear River lineage of today may be a remnant of a lower more broadly distributed Snake River Plain occupant. This is seen in three srYCT haplotypes (Figure 2) closely associated with the Bear River Bonneville cutthroat trout. These occurred in the Salt River and tributaries to Jackson Lake. It is also important to note that the main basin Bonneville cutthroat trout was too divergent from the Yellowstone cutthroat trout and Bear River Bonneville cutthroat trout lineages to connect to the TCS network as expected based on the phylogentic analysis (Figure 1). The TCS network supports the Bear River cutthroat trout being monophyletic with the Yellowstone cutthroat trout. Its current classification with the main Basin Bonneville cutthroat trout is a polyphyletic construction.

Greenback Cutthroat Trout Distribution

The greenback cutthroat trout's mitochondrial lineage is as divergent from the Colorado River cutthroat trout as are both the Bonneville and Rio Grande cutthroat trout (Figure 1). Yet the origin of the greenback cutthroat trout is thought to be from a recent invasion from the Colorado River (Behnke 1992, 2002). Until recently no genetic markers had been developed to clearly separate Colorado River cutthroat trout from greenback cutthroat trout. In fact, if the invasion had been as recent as the last glacial recession, it was likely that only rapidly evolving markers, such as microsatellites, could detect differences. Most genetic studies therefore focused on detecting both Yellowstone cutthroat trout and rainbow trout *O. mykiss* introgression into greenback cutthroat trout populations (e.g. Evans and Shiozawa 2001, 2002a, 2002b).

Metcalf et al. (2007) applied a suite of genetic markers to this problem and found that many greenback cutthroat trout populations contained evidence of Colorado River cutthroat trout introgression. Further, a Colorado River Basin population was identified as being greenback cutthroat trout. Other studies (Evans and Shiozawa, unpublished data) also identified Colorado River Basin greenback cutthroat trout populations. Greenback cutthroat trout in the Colorado River Basin could be due to translocation of fish from the Arkansas or South Platte rivers into the Colorado River Basin, or they could be relicts of a natural invasion of cutthroat trout across the continental divide.

If the greenback cutthroat trout are native to both sides of the continental divide, a haplotype network (Figure 3) should show divergence between the basins. Our results show that considerable divergence exists, with the three Arkansas Basin haplotypes being at least seven steps removed from Colorado River Basin haplotypes. This argues for relatively long isolation time between the westslope and eastslope greenback cutthroat trout. The four greenback cutthroat trout haplotypes in the Colorado River Basin are also at least as divergent from one-another as are the Arkansas haplotypes. Our tentative conclusion is that the Colorado River Basin greenback cutthroat trout is not indicative of transplanted stock originating from the Arkansas River Basin.

We suspect that the origin of the greenback cutthroat trout is from the Colorado River Basin, but it is also possible that the fish are from a Rio Grande cutthroat trout ancestor, that initially invaded the Arkansas River Basin and from there, the Colorado River Basin. A final interpretation of these data requires the addition of more populations and more individuals per population so that a better understanding of gene flow can be constructed. It may even be possible that highly diverse greenback cutthroat trout populations from the South Platte and the Arkansas basins were transferred to the Colorado River Basin generating the high diversity apparent in our haplotype network. If that is the case we would expect to find occasional evidence of no haplotype divergence between some Colorado River and Arkansas and South Platte individuals.

Another factor needs to be considered as well. If the greenback cutthroat trout did originate in the Colorado River Basin, then were Colorado River cutthroat trout also present when the greenback cutthroat trout invaded east across the continental divide? If the Colorado River cutthroat trout was present, then Colorado River cutthroat trout haplotypes may have also crossed the divide with the same invasion. If so, the Colorado River cutthroat trout mtDNA lineages currently found in the Arkansas and South Platte rivers may also be native to those basins. This can be tested by examining Colorado River cutthroat trout divergence patterns for similarities to what we see in the Greenback cutthroat trout. These relationships are currently under investigation in our laboratory.

Acknowledgments

Portions of this study were supported by the Carl and Victoria Sant Educational Endowment for a Sustainable Environment (to DKS), and funding from the Utah Division of Wildlife Resources, the Colorado Division of Wildlife, the Endangered Species Mitigation Fund of the Utah Department of Natural Resources, as well as numerous volunteers. All trout samples utilized in this study were obtained from tissues and DNA archived in the Fish Range of the Monte L. Bean Life Science Museum at BYU.

Literature Cited

Allendorf, F. W., and R. F. Leary. 1988. Conservation and distribution of genetic variation in a polytypic species, the cutthroat trout. Conservation Biology 2:170-184.

Behnke, R. J., 1992. Native trout of western North America. American Fisheries Society Monograph 6. Bethesda, Maryland.

Behnke, R. J., 2002. Trout and Salmon of North America. The Free Press. New York.

Billman, E. J., J. B. Lee, D. O. Young, M. D. McKell, R. P. Evans, and D. K. Shiozawa. 2010. Phylogenetic divergence in a desert fish: differentiation of speckled dace within the Bonneville, Lahontan, and Upper Snake River Basins. Western North American Naturalist 70:39-47.

Clement, M., D. Posada, and K. A. Crandall. 2000. TCS: a computer program to estimate gene genealogies. Molecular Ecology 9:1657-1659.

Evans, R. P., and D. K. Shiozawa. 2001. The genetic status of Greenback cutthroat trout (*Oncorhynchus clarki stomias*) populations in Colorado. Report to Colorado Division of Wildlife. Fort Collins, Colorado.

Evans, R. P., and D. K. Shiozawa. 2002a. The genetic status of Greenback cutthroat trout (*Oncorhynchus clarki stomias*) populations in Middle Fork Hayden Creek, South Apache Creek, Severy Creek, and Graneros Creek, Colorado. Report to Colorado Division of Wildlife, Fort Collins, Colorado.

Evans, R. P., and D. K. Shiozawa. 2002b. The genetic status of Greenback cutthroat trout (*Oncorhynchus clarki stomias*) populations in Bear Creek, North Taylor Creek, and Graneros Creek, Colorado. Report to Colorado Division of Wildlife, Fort Collins, Colorado.

Hall, T. A. 1999. BioEdit: a user-friendly biological sequence alignment editor and analysis program for Windows 95/98/NT. Nucleic Acids Symposium Series 41:95–98.

Houston, D. D., T. H. Ogden, M. F. Whiting, and D. K. Shiozawa. 2010. Polyphyly of the pikeminnows (Teleostei:Cyprinidae) inferred using mitochondrial DNA sequences. Transactions of the American Fisheries Society 139:303-315.

Hubbs, C. L., and R. R. Miller. 1948. The Great Basin, with emphasis on glacial and postglacial times: the zoological evidence. Bulletin of the University of Utah 38:17–166.

Kauwe, J. S. K., D. K. Shiozawa, and R. P. Evans. 2004. Phylogeographic and nested clade analysis of the stonefly *Pteronarcys californica* (Plecoptera:Pteronarcyidae) in Western USA. Journal of the North American Benthological Society 23:824-838.

Loudenslager E. J., and G. A. E. Gall. 1980. Geographic patterns of protein variation and subspeciation in cutthroat trout, *Salmo clarki*. Systematic Zoology 29:27-42.

Martin, M. A., D. K. Shiozawa, E. J. Loudenslager, and J. N. Jensen. 1985. An electrophoretic study of cutthroat trout populations in Utah. Great Basin Naturalist 45:677-687.

Metcalf, J. L., V. L. Pritchard, S. M. Silvestri, J. B. Jenkins, J. S. Wood, D. E. Cowley, R. P. Evans, D. K. Shiozawa, and A. P. Martin. 2007. Across the great divide: genetic forensics reveals misidentification of endangered cutthroat trout populations. Molecular Ecology 16:4445-4454.

Pritchard, V. L., J. L. Metcalf, K. Jones, A. P. Martin, and D. E. Cowley. 2009. Population structure and genetic management of Rio Grande cutthroat trout (*Oncorhynchu clarkii virginalis*). Conservation Genetics 10:1209-1221.

Smith, G. R., T. E. Dowling, K. W. Gobalet, T. Lugaski, D. K. Shiozawa, and R. P. Evans. 2002. Biogeography and timing of evolutionary events among Great Basin fishes. Pages 175-234 *in* R. Hershler, D. B. Madsen and D. R. Currey, editors. Great Basin Aquatic Systems History. Smithsonian Contributions to the Earth Sciences, number 33. Smithsonian Institution Press, Washington, DC.

Smith, G. R. 1981. Late Cenozoic freshwater fishes of North America. Annual Review of Ecology and Systematics 12:163–193.

Stamatakis, A. 2006a. RAxML-VI-HPC: Maximum Likelihood-based phylogenetic analyses with thousands of taxa and mixed models. Bioinformatics 22:2688–2690.

Stamatakis, A. 2006b. Phylogenetic models of rate heterogeneity: a high performance computing perspective. In: Proceedings of the 20th IEEE/ACM International Parallel and Distributed Processing Symposium (IPDPS2006), Rhodos, Greece.

Stamatakis A., P. Hoover, and J. Rougemont. 2008. A rapid bootstrap algorithm for the RaxML web-servers. Systematic Biology 75:758–771.

Stutz, H. L., D. K. Shiozawa, and R. P. Evans. 2010. Inferring dispersal of aquatic invertebrates from genetic variation: a comparative study of an amphipod and mayfly in Great Basin Springs. Journal of the North American Benthological Society 29:1132-1147.

Tamura, K., J. Dudley, M. Nei, and S. Kumar. 2007. MEGA4: molecular evolutionary genetics analysis (MEGA) software version 4.0. Molecular Biology and Evolution 24:1596–1599.

Taylor, D. W. 1985. Evolution of freshwater drainages and mollusks in Western North America. Pages 265–321 *in* C. J. Smiley (editor). Late Cenozoic history of the Pacific Northwest. Pacific Division of the American Association for the Advancement of Science, San Francisco, California.

Templeton, A. R., K. A. Crandall, and C. F. Sing. 1992. A cladistic analysis of phenotypic associations with haplotypes inferred from restriction endonuclease mapping and DNA sequence data. I.

Base theory and analyses of alcohol dehydrogenase activity in *Drosophila*. Genetics 117:343–351.

Wilson, W. D., and T. F. Turner. 2009. Phylogenetic analysis of the Pacific cutthroat trout (*Oncorhynchus clarkii* ssp.: Salmonidae) based on partial mtDNA sequences: a closer look at the highly fragmented inland species. Molecular Phylogenetics and Evolution 521:406-415

Session 4: Wild Trout in the Face of invasive Species and Diseases

THE INFLUENCE OF *DIDYMOSPHENIA GEMINATA* ON FISHERIES RESOURCES IN RAPID CREEK, SOUTH DAKOTA – AN EIGHT YEAR HISTORY

D. A. James and S. R. Chipps
U.S. Geological Survey South Dakota Cooperative Fish and Wildlife Research Unit, Department of Wildlife and Fisheries Sciences, South Dakota State University, Brookings, South Dakota 57007, USA.
Daniel.James@sdstate.edu; Steven.Chipps@sdstate.edu

ABSTRACT – The aquatic nuisance diatom *Didymosphenia geminata* was established in Rapid Creek in the Black Hills of South Dakota in 2002. Shortly thereafter, large declines (>50%) of the naturalized brown trout *Salmo trutta* population were observed. We evaluated the influence of water resources and *D. geminata* on (1) declines in brown trout biomass, (2) changes in food resources, and (3) diet of brown trout in Black Hills streams. Drought conditions were largely responsible for trout declines in Black Hills streams. However, comparison of brown trout size-structure between the pre-*D. geminata* and post-*D. geminata* periods revealed that juvenile brown trout abundance increased while adult abundance decreased in Rapid Creek. Changes in food resources in *D. geminata*-impacted areas were thought to favor juvenile brown trout and negatively impact adults. In the presence of *D. geminata*, macroinvertebrate abundance was composed of fewer, larger taxa and higher numbers of smaller taxa (i.e., chironomids). Brown trout in Rapid Creek consumed fewer ephemeropterans and a high amount of dipterans. Nonetheless, diet analysis showed that brown trout in Rapid Creek consumed as much or more prey than trout from two other streams unaffected by *D. geminata*. Moreover, relative weight of brown trout from Rapid Creek was high (>100), implying that food availability was not limiting. These findings imply that *D. geminata* did not negatively impact feeding and condition of brown trout in Rapid Creek, although mechanisms affecting size-structure in Rapid Creek remain unknown.

INTRODUCTION

The spread and establishment of *Didymosphenia geminata* has prompted much concern in North America and New Zealand (Branson 2006; Kilroy 2004; Spaulding and Elwell 2007). It is capable of producing large masses of extracellular stalks that can cover up to 100% of the stream bottom in areas of high infestation, which can make *D. geminata* populations a nuisance in stream ecosystems. Recent research on invertebrate communities has shown that invertebrate composition tends to shift from larger taxa (i.e., Ephemeroptera, Plecoptera, Trichoptera [EPT]) to smaller taxa such as Diptera in areas impacted by *D. geminata* (Gillis and Chalifour 2009; Kilroy et al. 2009; James et al. 2010b). Total invertebrate abundance tends to increase in areas where *D. geminata* is present (Gillis and Chalifour 2009;Kilroy et al. 2009). *D. geminata* was first documented in the Black Hills of South Dakota in 2002 and became established concurrent with drought conditions (2000-2008). Shortly after the appearance of *D. geminata* in Rapid Creek, large biomass declines (>50%) of the naturalized brown trout *Salmo trutta* population in Rapid Creek were observed. It was unclear if drought conditions or the presence of *D. geminata* were responsible for brown trout biomass declines. Here, we evaluate the influence of water resources and *D. geminata* on (1) declines in brown trout biomass, (2) alteration of food resources, and (3) diet of brown trout in Black Hills streams.

METHODS

The various components of our research were conducted within four stream reaches in South Dakota's Black Hills: Spearfish Creek, an unregulated stream that flows through Spearfish Canyon (*D. geminata* absent), upper Rapid Creek (tailwater reach below Pactola Reservoir; *D. geminata* present), lower Rapid Creek (in Rapid City below Canyon Lake; *D. geminata* absent), and Castle Creek (tailwater reach below Deerfield Reservoir; *D. geminata* absent; Figure 1). For a detailed description of the Rapid and Spearfish Creek study reaches, see James et al. (2010a, b).

We estimated *D. geminata* biovolume (when stream flows permitted) once per month from March through September (high April discharge prohibited field sampling) in each study section from 2007-2009 using an approach modified from Hayes et al. (2006) and Kilroy et al. (2006). For each of one hundred randomly selected rocks from a standard riffle at each sampling site, percent coverage of *D. geminata* was visually estimated and the thickness of the *D. geminata* mat was measured (mm). Thickness was assigned a score from 0 to 5 based on the following: 0; 1 (< 1 mm thick); 2, (1-5 mm); 3, (6-15 mm); 4 (16-30 mm); 5, (> 30 mm). The percent coverage of *D. geminata* was multiplied by the thickness score to provide a *D. geminata* biovolume index (DBI), which ranged from 0 to 500.

We examined water resources from 2000 to 2007. Since 2000, annual precipitation in the Black Hills region has generally been below average, leading to an extended drought period that lasted until fall 2008. To characterize periods of relatively higher and lower water availability from 2000-2007, we evaluated mean monthly stream discharge and mean monthly summer (June-August) stream temperature from two time periods, early-drought (2000-2002) and late-drought (2005-2007) using a paired t-test ($\alpha \leq 0.05$) to verify that mean monthly discharge was indeed lower during the late-drought than the early-drought period (see James et al. 2010a).

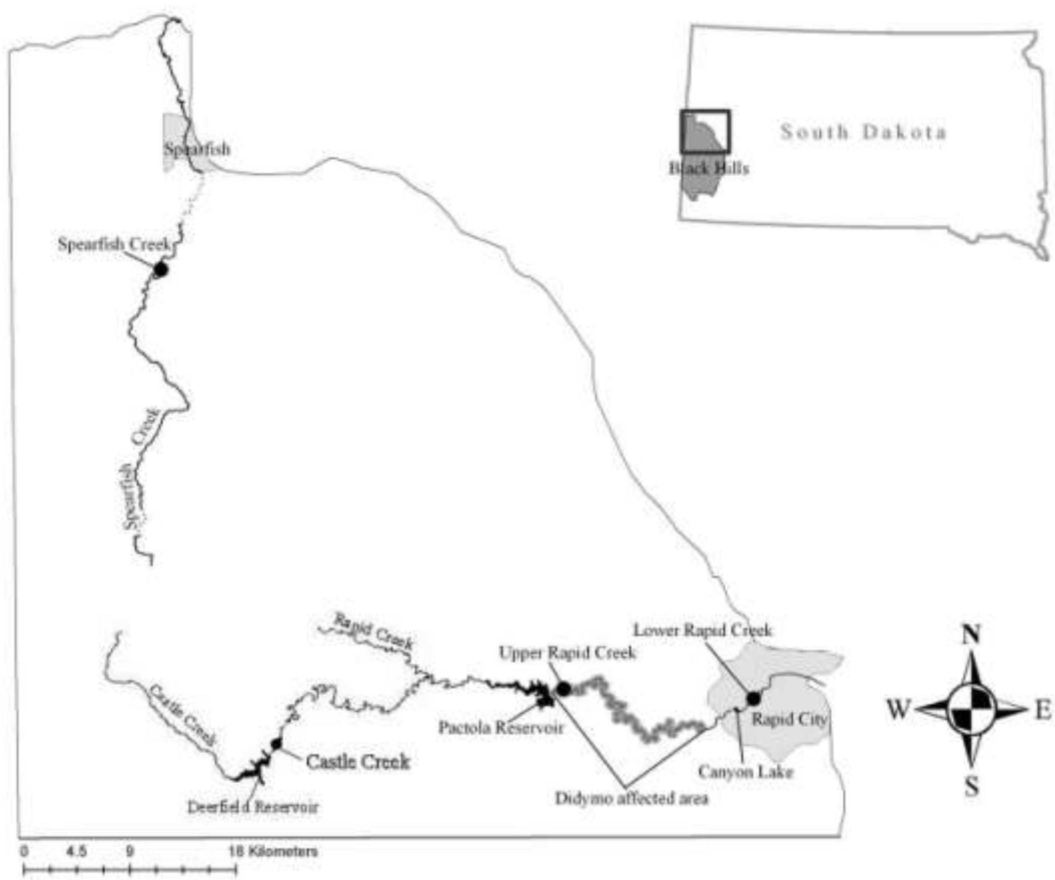


Figure 1. Locations of the Spearfish Creek, lower Rapid Creek, upper Rapid Creek, and Castle Creek study reaches in the Black Hills, South Dakota. The current *D. geminata* distribution in Rapid Creek is indicated by dark shading.

Next, we analyzed brown trout biomass and size structure in our study reaches. Brown trout were sampled by multiple-pass depletion backpack electrofishing surveys in the fall (late-August through September) in 2000-2002 and 2005-2007 at standardized locations from Rapid and Spearfish creeks. Population and biomass estimates were calculated for each year sampled in each stream. Brown trout in this study were assigned to one of two size categories. Fish ≤ 199 mm TL were considered juveniles and fish ≥ 200 mm TL were considered adults. Relative weight was calculated by dividing the weight of each brown trout by its length-specific standard weight (Anderson and Neumann 1996). We used a repeated measures analysis of variance (RMANOVA) to test for differences in mean juvenile and adult biomass between early- and late-drought time periods (PROC MIXED, SAS 9.1). Similarly, we used a RMANOVA to compare size structure (i.e., ratio of juvenile to adults) of brown trout between the early- and late-drought time periods in each stream reach ($\alpha \leq 0.05$) (Neumann and Allen 2007; see James et al. 2010a).

To examine the abundance and composition of macroinvertebrates in Rapid Creek, we selected four sites to sample – two in areas with high relative abundance of *D. geminata* and two with low relative abundance. At each of the four sites, benthic invertebrates were collected using a D-frame dip net, a Surber sampler, and drift nets. Invertebrate sampling was conducted in September and October 2006. Invertebrates were identified to Order. We tested for differences among the four sites using multivariate analysis of variance (MANOVA; SAS 9.1 SAS Institute 2007). We also calculated the proportion of EPT for each sampling gear at each site. Similarly, the proportion of dipterans was calculated. We tested for differences among sites for EPT and Diptera using analysis of variance (see James et al. 2010b).

Finally, we sampled diets of brown trout from Spearfish, Rapid, and Castle creeks using gastric lavage monthly from June through August 2008 - 2009. From each sampling occurrence we collected up to 10 brown trout in three size categories (100-199, 200-299, and >300 mm TL). Stomach contents were preserved in ethanol, enumerated, identified to Order, and weighed (dry weighting) to quantify biomass. We compared gut contents of brown trout among streams using mean percent composition by weight (MWi; Chipps and Garvey 2007) of the most common invertebrate orders using analysis of variance (ANOVA; data were $\arcsin\sqrt{p}$ transformed prior to analysis). Alpha was set at ≤ 0.05 and a Bonferroni correction was used; a Tukey test was used to evaluate differences among streams. We also calculated a gut fullness index by dividing the weight of the prey in the stomach by the weight of the fish and used analysis of covariance (ANCOVA) with length as a covariate ($\alpha \leq 0.05$). Finally, we conducted a weight-at-length (condition) analysis (using ANCOVA with fish length as a covariate to control for effects of differing size ranges; data were log transformed prior to analysis; $\alpha \leq 0.05$; Pope and Kruse 2007).

Results

The Spearfish and Rapid creeks study sections had significantly lower mean monthly discharges during the late-drought compared to the early-drought (Spearfish Creek, $t_{11} = 4.42$, $P = 0.001$; lower Rapid Creek, $t_{11} = 6.24$, $P < 0.0001$; upper Rapid Creek, $t_{11} = 4.02$, $P = 0.002$; Table 1). In contrast to stream discharge, mean summer stream temperature did not differ significantly between the early- and late-drought time periods in each study reach (Spearfish Creek, $t_2 = 0.86$, $P = 0.48$; lower Rapid Creek, $t_2 = 0.21$, $P = 0.85$; upper Rapid Creek, $t_2 = 0.03$, $P = 0.97$; Table 1; see James et al. 2010a).

Mean *D. geminata* biovolume in upper Rapid Creek was variable from March to September during 2007-2009 (Figure 2). April values were not obtained due to high stream discharge. Mean DBI was 57.6 (SE = 6.8), and mean substrate coverage percentage was 24.2 (SE = 3.0). Visible *D. geminata* was absent from the Castle and Spearfish creeks.

Mean biomass for adult brown trout in all three stream sections was significantly lower in the late-drought than the early-drought (Spearfish Creek, $P = 0.02$; lower Rapid Creek, $P = 0.01$; upper Rapid Creek, $P = 0.01$; Table 1). For juvenile brown trout in lower Rapid Creek, mean biomass was significantly lower during the late-drought time period ($P = 0.01$; Table 1). In Spearfish Creek, juvenile biomass was not significantly different between time periods ($P = 0.14$). Juvenile biomass in upper Rapid Creek was also not significantly different ($P = 0.08$; Table 1), but in contrast to the other two study reaches, juvenile brown trout biomass increased in upper Rapid Creek (see James et al. 2010a).

Table 1. Mean summer (June – August) stream temperature (^oC), mean annual monthly discharge ($m^3 \cdot s^{-1}$), and mean biomass (kg/ha) of brown trout in Spearfish Creek, upper Rapid Creek, and lower Rapid Creek during early- (2000-2002) and late-drought (2005-2007) time periods in the Black Hills, South Dakota. Values in parentheses represent 1 S.E. Adapted from James et al. (2010a).

	Temperature		**Discharge**		**Adult Biomass**		**Juvenile Biomass**	
Stream	Early	Late	Early	Late	Early	Late	Early	Late
Spearfish	12.4 (0.5)	11.5 (0.5)	1.95 (0.08)	1.50 (0.14)	238 (24)	69 (29)	43 (7)	23 (8)
Upper Rapid	9.8 (1.2)	9.8 (0.6)	1.41 (0.15)	0.84 (0.17)	159 (17)	32 (17)	14 (18)	73 (18)
Lower Rapid	19.2 (0.8)	19.3 (0.2)	2.01 (0.19)	0.94 (0.11)	272 (27)	91 (27)	136 (13)	45 (13)

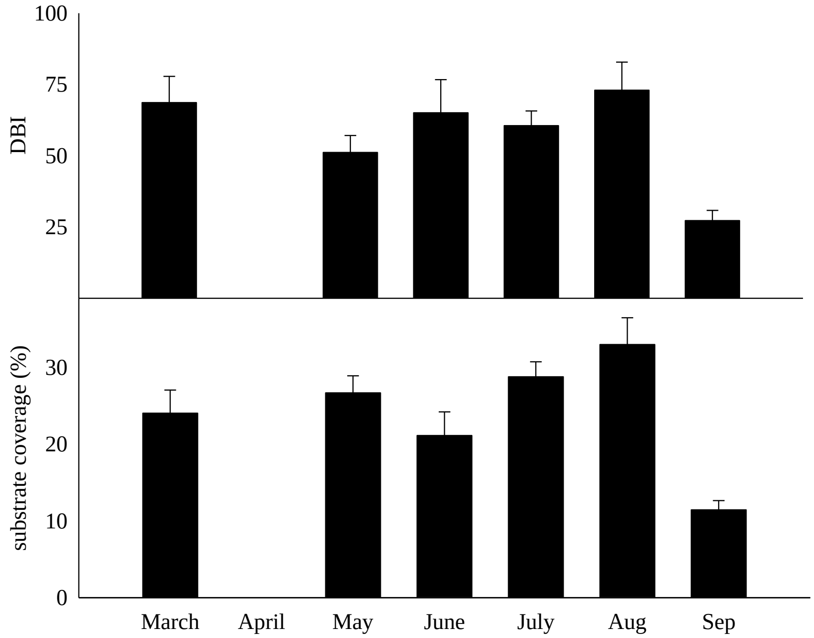


Figure 2. Three-year monthly mean *D. geminata* biovolume index (DBI) and percent substrate coverage in Rapid Creek from March through September 2007-2009. Bars represent 1 SE. No data were available for April.

Representatives were collected from several orders of insects, but because EPT and Diptera represented 72 to 94% of the insects collected at each site, we focused our analysis on those four orders. Invertebrate abundance varied significantly among locations for each of the gear types used (MANOVA: dip nets, $F_{12, 35} = 2.05$, $P = 0.04$; Surber, $F_{12, 13} = 4.32$, $P = 0.006$; drift nets, $F_{12, 34} = 4.25$, $P = 0.004$). For each gear type used, Diptera abundance varied significantly among locations and was generally higher at locations with *D. geminata*. The proportion of EPT varied among locations and was generally higher at sampling locations without *D. geminata*. In contrast, the percentage of Diptera was higher at sites with *D. geminata* as indexed by Surber samples ($F_{3, 17} = 14.2$, $P < 0.0001$) and drift nets ($F_{3, 8} = 14.46$, $P = 0.0014$; see James et al. 2010b).

We analyzed the gut contents of 316 brown trout collected from Castle, Spearfish and Rapid creeks from June through August in 2008 and 2009. Prey items (n = 20,615) representing 19 Orders were used in the analyses. The most common prey items encountered in stomach samples were from the Orders Ephemeroptera, Plecoptera, Trichoptera, Diptera, and Amphipoda. All other prey items were combined and referred to as other. We observed significant differences in mean percentage composition by weight (MWi) throughout the study period (Table 2). The Ephemeroptera, Diptera, Amphipoda, and other Orders had significant differences in the summer time period (Table 2). Analysis of gut fullness (g prey/g of predator) revealed that brown trout from Rapid and Castle creeks had more prey biomass in their stomach compared with brown trout from Spearfish Creek ($F_{3, 306} = 4.18$, $P = 0.0161$; Figure 3). The interaction term ($F_{5, 304} = 1.76$; $P = 0.1733$) indicated that fish had similar trends in gut fullness relative to length in all three study streams. Relative weights of brown trout were highest in Rapid Creek, followed by brown trout in Castle and Spearfish creeks ($F_{3, 315} = 20.58$; $P < 0.0001$). The interaction term (F 2, 313 = 0.70; $P = 0.4990$) indicated that fish from each stream had similar trends in weight relative to length. Relative weights were generally higher in Rapid Creek compared to the other two study sections (Figure 3).

Table 2. Mean percent composition by dry weight (MWi; g) and standard error of gut contents from brown trout in Rapid, Castle, and Spearfish creeks, South Dakota. Results of ANOVA analyses. The summer period represents pooled data from June to August 2008-2009. Values with the same letters are not significantly different (*P* > 0.0083).

		Stream							
	Time	Castle		Rapid		Spearfish			
Order	Period	MWi	SE	MWi	SE	MWi	SE	F	P
Ephemeroptera	summer	0.443 [a]	0.03	0.263 [b]	0.04	0.546 [a]	0.06	11.35	< 0.0001
Plecoptera	summer	0.055	0.01	0.049	0.01	0.033	0.01	0.54	0.5862
Trichoptera	summer	0.406	0.03	0.300	0.03	0.326	0.05	2.80	0.0624
Diptera	summer	0.238 [a]	0.02	0.461 [b]	0.05	0.448 [b]	0.04	13.55	< 0.0001
Amphipoda	summer	0.450 [a]	0.04	0.490 [a]	0.05	0.035 [b]	0.02	24.13	< 0.0001
Other	summer	0.234 [a]	0.03	0.160 [a]	0.03	0.394 [b]	0.06	9.78	< 0.0001

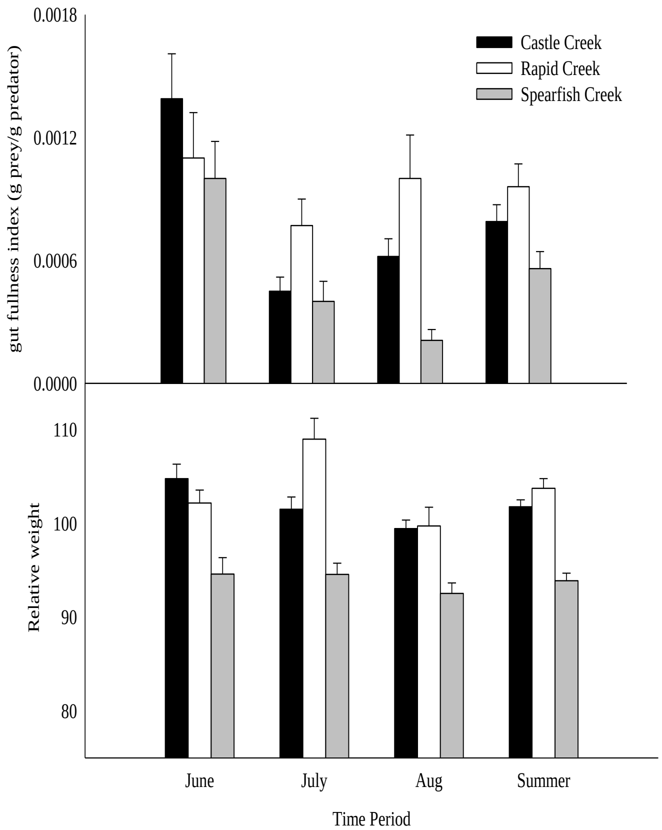


Figure 3. Mean gut fullness index (g prey / g predator) and mean relative weight of brown trout from Castle, Rapid, and Spearfish creeks, South Dakota. The summer period represents pooled data from June-August 2008-2009. Bars represent SE.

Discussion

Since the establishment of *D. geminata* in Rapid Creek, the naturalized brown trout population has experienced a large (> 50%) biomass decline. Initially, declines in biomass were attributed to *D. geminata* due to an incomplete understanding of the diatom and its interactions with fish. We determined that drought conditions were largely responsible for overall trout biomass decreases, regardless of the presence of *D. geminata* (James et al. 2010a). However, comparison of brown trout size-structure between the early-drought (pre-*D. geminata*) and late-drought (post-*D. geminata*) periods revealed that juvenile brown trout abundance increased while adult abundance decreased in Rapid Creek (James et al. 2010a). Reasons for these size-structure differences were unknown, but changes in food resources in *D. geminata*-impacted Rapid Creek were suspected.

Changes in invertebrate abundance and composition have been documented in recent studies. Invertebrate composition tends to shift from larger taxa (i.e., EPT) to smaller taxa such as Diptera in areas impacted by *D. geminata*, while total invertebrate abundance also generally increases (Larson 2007; Gillis and Chalifour 2009; Kilroy et al. 2009; James et al. 2010b). A higher abundance of dipterans and lower percentage of EPT taxa were present in *D. geminata*-impacted areas of Rapid Creek compared with non-impacted areas (James et al. 2010b). An increase in numbers of smaller invertebrate Diptera taxa (e.g., Chironomidae) and a decrease in number of larger, energy-rich EPT taxa could explain increased numbers of juvenile brown trout in Rapid Creek (i.e., increased size-specific food abundance). Food resources for juvenile brown trout were abundant while these same food resources could be limiting for adult brown trout growth and survival.

Examination of brown trout gut contents from upper Rapid, Castle, and Spearfish creeks, showed a lower composition of ephemeropterans in brown trout from Rapid Creek (*D. geminata* present; Table 2). Composition of plecopterans and trichopterans was not different in Rapid Creek compared with Castle and Spearfish creeks (*D. geminata* absent). Brown trout in Rapid Creek consumed a high composition of dipterans as well (Table 2). These findings were consistent for both juvenile and adult brown trout, which supported our hypothesis that changes in invertebrate composition may have influenced decreases in adult biomass. However, after analysis of gut fullness index, we observed that brown trout from Rapid Creek consumed more prey overall than brown trout in Castle or Spearfish creeks (Figure 3). Moreover, relative weight of brown trout from Rapid Creek was generally high (>100), implying that food availability was not limiting. Although brown trout in *D. geminata* affected Rapid Creek consumed fewer ephemeropterans and a high amount of lower energy-density prey items (i.e., dipterans) compared to the non-impacted streams, the brown trout also consumed a high amount of energy-rich Amphipods. Despite differences in prey consumption among *D. geminata* affected and unaffected streams, brown trout in Rapid Creek (*D. geminata* affected) were able to consume enough prey such that food resources, although altered, were not limiting.

Our findings imply that despite changes in invertebrate composition, *D. geminata* (at relatively low levels; approximately 25% substrate coverage, < 5mm thick) did not negatively impact gut fullness or condition of brown trout in Rapid Creek. Further research is necessary to determine if *D. geminata* negatively affects trout prey consumption in higher levels of *D. geminata* coverage and biovolume. Furthermore, more research is necessary to determine the mechanisms affecting size-structure differences in Rapid Creek.

Acknowledgements

We thank J. Wilhite and staff from the South Dakota Department of Game, Fish & Parks and South Dakota State University for laboratory and field assistance during this study. Funding for this project was provided by Federal Aid in Sport Fish Restoration (Project F-15-R 1514) administered by the South Dakota Department of Game, Fish and Parks. Any use of trade names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References

Anderson, R. O. and R. M. Neumann. 1996. Length, weight, and associated structural indices. Pages 447-482 *in* B. R. Murphy and D. W. Willis, editors. Fisheries Techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Branson, Johannah. 2006. *Didymosphenia geminata* economic impact assessment. New Zealand Institute of Economic Research, Wellington.

Chipps, S. R. and J. E. Garvey. 2007. Assessment of diets and feeding patterns. Pages 473-514 *in* C. S. Guy and M. L. Brown, editors. Analysis and interpretation of freshwater fishes data. American Fisheries Society, Bethesda, Maryland.

Gillis, C., and M. Chalifour. 2009. Changes in the macrobenthic community structure following the introduction of the invasive algae *Didymosphenia geminata* in the Matapedia River (Quebec, Canada). Hydrobiologia 647: 63-70.

Hayes, J. W., K. Shearer, and J. Hay. 2006. Invertebrate drift and trout growth potential in a Didymo (*Didymosphenia geminata*) affected reach of the Mararoa River. Prepared for Biosecurity New Zealand. Cawthorn Report 1178. Nelson, New Zealand.

James, D. A., J. W. Wilhite, and S. R. Chipps. 2010a. Influence of drought conditions on brown trout biomass and size structure in the Black Hills, South Dakota. North American Journal of Fisheries Management 30: 791-798.

James, D. A., S. H. Ranney, S. R. Chipps and B. D. Spindler. 2010b. Invertebrate composition and abundance associated with *Didymosphenia geminata* in a montane stream. Journal of Freshwater Ecology 25: 235-241.

Kilroy, Cathy. 2004. A new alien diatom, *Didymosphenia geminata* (Lyngbye) Schmidt: its biology, distribution, effects and potential risks for New Zealand fresh waters. National Institute of Water and Atmospheric Research, Project ENS05501, Report CHC2004-128, Christchurch, New Zealand.

Kilroy, C., Lagerstedt, A., Davey, A., Robinson, K. 2006. Studies on the survivability of the exotic, invasive diatom *Didymosphenia geminata* under a range of environmental and chemical conditions. National Institute of Water and Atmospheric Research, Project MAF06506, Report CHC2006-116, Christchurch, New Zealand.

Kilroy, C., S. T. Larned, and B. J. F. Biggs. 2009. The non-indigenous diatom *Didymosphenia geminata* alters benthic communities in New Zealand rivers. Freshwater Biology 54: 1990-2002.

Larson, A. M. 2007. Effects of nuisance blooms of *Didymosphenia geminata* on benthic community composition in Rapid Creek, South Dakota. South Dakota Department of Environment and Natural Resources. Biological Assessment Interim Report. Pierre.

Neumann, R. M., and M. S. Allen. 2007. Size structure. Pages 375-421 *in* C. S. Guy and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

Pope, K. L. and C. G. Kruse. 2007. Condition. Pages 423-471 *in* C. S. Guy and M. L. Brown, editors. Analysis and interpretation of freshwater fishes data. American Fisheries Society, Bethesda, Maryland.

Spaulding, S., and L. Elwell. 2007. Increase in nuisance blooms and geographic expansion of the freshwater diatom *Didymosphenia geminata*: recommendations for response. U.S. Environmental Protection Agency, Denver, Colorado, and Federation of Flyfishers, Livingston, Montana.

EXAMINING THE OCCURRENCE OF WILD RAINBOW TROUT IN THE BRAS D'OR LAKES, NOVA SCOTIA: USING SCALE PATTERN ANALYSIS TO DIFFERENTIATE HATCHERY AND WILD POPULATIONS

Reginald J. Madden*, John L. MacMillan

Nova Scotia Department of Fisheries and Aquaculture, Inland Fisheries Division, Fisheries Management Section, 91 Beeches Road, Pictou, Nova Scotia, Canada, B0K 1H0, maddenrj@gov.ns.ca

Joseph Apaloo

St. Francis Xavier University, Department of Mathematics, Statistics and Computing Science, P.O. Box 5000, Antigonish, Nova Scotia, Canada, B2G 2W5

ABSTRACT — Rainbow trout *Oncorhynchus mykiss*, a known invasive species, are the second most popular sport fish in Nova Scotia. First introduced in 1899, the successful reproduction of rainbow trout was believed to be limited and angler catches were dependant on stocking. Wild juvenile rainbow trout electrofished from the Bras d'Or Lakes watershed were directly compared to unknown (angled) and known origin (hatchery) rainbow trout.

Rainbow trout reared under aquaculture conditions experience rapid growth resulting in morphological differences from wild juveniles. Mean length and weight at age 1+ of known origin wild and hatchery trout were 13.1 cm (29.5 g) and 27.6 cm (252.8 g), respectively. Scale analysis revealed hatchery samples had more circuli spaced farther apart. Significant variations were detected in the mean difference in distance (12.3 µm) of the first six circuli pairs of wild and hatchery rainbow trout (T-test, $P<0.05$). A Stepwise Logistic Regression Model was constructed to classify the unknown origin samples into either wild or hatchery. Approximately 70% of the known origin data were randomly selected to construct the logistic model. The remaining 30% were used for validation. The dependent variable was wild or hatchery and the predictor variables were weight, length and the six circuli pair distances. The significant predictor was the 6th circuli pair distance. The model classified 71% of the rainbow trout with unknown origin as hatchery and 29% as wild origin suggesting that rainbow trout reproducing in the wild contribute to a significant proportion of the angler catch.

Rainbow trout are successfully reproducing in Nova Scotia and this is the first research to confirm successful reproduction using scale analysis. All aquaculture operations are encouraged to use triploid rainbow trout to reduce impacts associated with successful reproduction on native populations of brook trout *Salvelinus fontinalis* and Atlantic salmon *Salmo salar*.

INTRODUCTION

The native range of the rainbow trout *Oncorhynchus mykiss* is the Eastern Pacific Ocean and freshwater lying west of the Rocky Mountains, extending from northwestern Mexico to the Kuskokwim River in Alaska (Scott and Crossman 1998). Rainbow trout were first introduced to Nova Scotia in 1899 (Scott and Crossman 1998). They are true spring spawners, preferring smaller tributaries of rivers but will also use suitable environments within lakes. They tend to inhabit open, fast flowing regions of streams similar to native Atlantic salmon *Salmo salar* parr (Cunjak and Green 1983) and are more tolerant of warm water compared to native brook trout *Salvelinus fontinalis* (Cunjak and Green 1986). In other areas of North America, rainbow trout exhibit invasive characteristics towards brook trout (Moore et al. 1983; Larson and Moore 1985; Larson et al. 1995) and other salmonids (McKenna and Johnson 2005; Boyer et al. 2008; Metcalf et al. 2008; Thibault et al. 2009). Rainbow trout are one of the most acid-sensitive salmonids (Chadwick and Bruce 1981) and acidic conditions in mainland Nova Scotia likely impede successful reproduction of rainbow trout and, therefore, their significance as an invasive species. Past electrofishing surveys in a number of tributaries of the Bras d'Or Lakes have detected the presence of juveniles and suggest that limited natural reproduction has occurred (Sabean 1983).

In Nova Scotia, rainbow trout have a long history of being grown commercially in aquaculture facilities and stocked from hatcheries to create sportfishing opportunities. The two main known contributors to wild rainbow trout populations in Nova Scotia are aquaculture escapement and hatchery stocking. Development of a reliable method to distinguish between hatchery and wild populations of rainbow trout will aid in future management decisions for this popular sport fish, and help to develop a greater understanding of their biological parameters. The contribution of wild origin rainbow trout to the angler creel provides insight as to their importance recreationally and as competitors with wild native stocks.

Rainbow trout are the second most popular sport fish in Nova Scotia (Fisheries and Oceans 2007). The most productive rainbow fisheries are located within the Bras d'Or Lakes and anglers have reported that catches of large ripe female rainbow trout are common in the Fall while angling historic Atlantic salmon rivers. Although rainbow trout are found in tidal waters, rivers and lakes throughout the province, little research has been conducted to examine their populations and how they adapt to Nova Scotia's environment and impact native species. Although mixed-sex rainbow trout were stocked prior to 2007, a common inference, has been that rainbow trout have limited reproductive success in Nova Scotia. Wild rainbow trout referred to in this paper are the offspring of naturally spawning hatchery or aquaculture fish.

Scale patterns show the spacing and number of circuli on the scale, which are correlated with food consumption and growth (Bilton and Robins 1971). Rainbow trout within a hatchery environment experience feeding rates that are characteristically much greater than those experienced in the wild, especially in the first year of life. Scale pattern analysis has been successfully used to differentiate between hatchery and wild salmonids by analyzing the freshwater growth zone and the location of the first annulus. Quantitative scale pattern analysis techniques have been developed to differentiate between farmed and wild Atlantic salmon in the maritime provinces (Stokesbury et al. 2001) but currently, no quantitative scale analysis technique exists to estimate proportions within mixed populations of introduced wild and hatchery rainbow trout. Quantitative scale pattern analysis methods independent of age determination are essential to avoid inclusion of false annuli on hatchery rainbow trout scales (Bernard and Myers 1996; Davis and Light 1985) and other potential visual pattern identification errors. Relatively low agreement between fisheries biologists on visual determinations of freshwater age of rainbow trout (Davis and Light 1985) further suggests that quantitative techniques independent of freshwater age could improve the accuracy of population estimates. If successful, this quantitative scale pattern analysis method can, regardless of accurate annulus confirmation (aging), calculate (with degrees of precision) the proportion of wild and hatchery rainbow trout in a mixed population while greatly reducing potential inter-observer analysis error.

The purpose of this paper is threefold: (1) to confirm that rainbow trout are successfully reproducing in Nova Scotia, (2) to determine if scale pattern analysis can be used with a high degree of confidence, to differentiate between hatchery and wild rainbow trout and (3) to estimate the proportion of wild rainbow trout in the angler catch from the Bras d'Or Lakes watershed.

STUDY AREA

The study area focused on Nova Scotia's largest estuary, the Bras d'Or Lakes, Cape Breton Island, Nova Scotia, Canada (Figure 1). The Bras d'Or Lakes watershed has an area of 3,565 km^2 with 12 sub-watersheds and approximately 1,000 km of coastline (Krauel 1976; Parker et al 2007). The Bras d'Or Lakes fall within the boundaries of all four counties (Cape Breton, Inverness, Richmond, Victoria) and cover approximately one-third of Cape Breton Island. Forty-two percent of all freshwater that flows into the Bras d'Or Lakes originates from six major rivers. The drainage basin area of each of

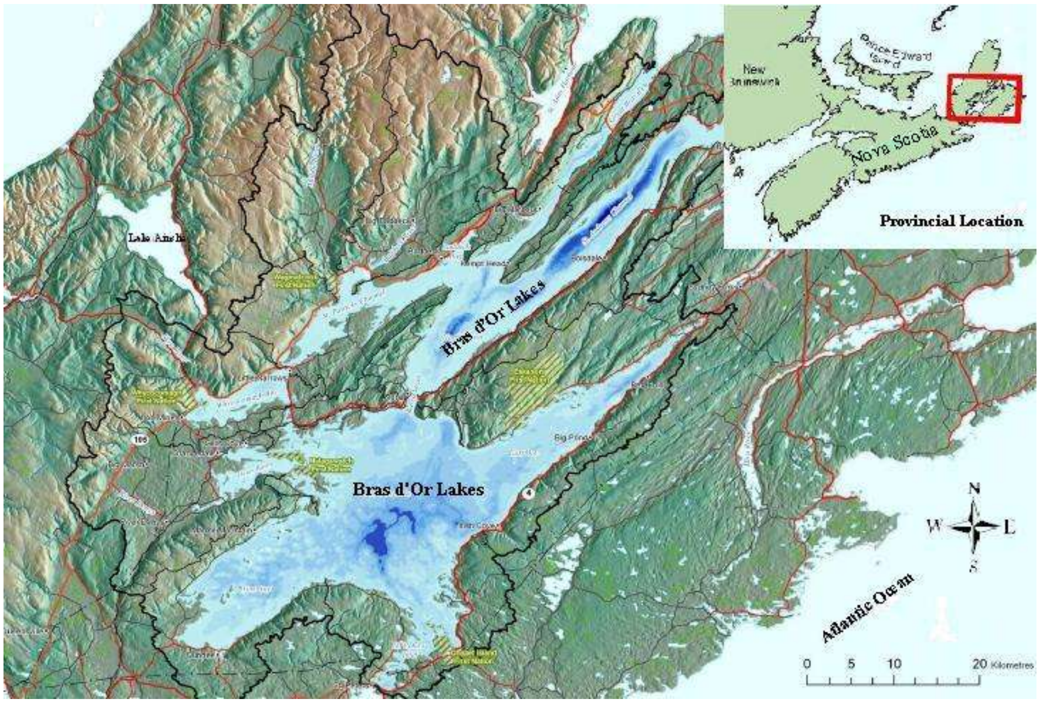


Figure 1. The study area: Bras d'Or Lakes, Nova Scotia, Canada. Map inlay depicts the provincial location of the study area.

these rivers is: Middle (325 km^2), Baddeck (295 km^2), Denys (215 km^2), Skye (120 km^2), Black (42 km^2), and Washabuck (23 km^2). Smaller systems account for the remaining 58% of flowage (Krauel 1976; Parker et al 2007).

METHODS

Three different samples of rainbow trout were collected during 2008 and 2009. Staff from the Nova Scotia Department of Fisheries and Aquaculture (NSDFA) – Inland Fisheries Division, electrofished seven lotic systems in the Bras d'Or Lakes watershed to assess the presence or absence of naturally reproducing rainbow trout; Skye River, Breac's Brook, McNabs Brook, Irish Cove Brook, MacRae Brook, North Branch Baddeck River, and New Glen Brook. Although River Denys, also part of the Bras d'Or Lakes watershed, was not specifically evaluated for rainbow trout, intensive trapping and electrofishing conducted simultaneous to this study did not reveal the presence of juvenile rainbow trout. Based on the timing of stocking occurrences and size and weight at age, juvenile samples electrofished from the Bras d'Or Lakes watershed were presumed the progeny of successful wild spawning. Rainbow trout collected from Fraser's Mills Hatchery were used for analysis and comparison to wild rainbow trout. Fraser's Mills Hatchery is operated by the NSDFA and is located in Antigonish County, Nova Scotia. Anglers provided the third sample of fish, angled throughout the Bras d'Or Lakes and its tributaries. The specific origin (hatchery or wild) of these samples was unknown.

A single pass technique with a Smith-Root backpack model LR-24 was used for electrofishing the lotic systems. Areas of assessment on each system were either chosen randomly or based on vicinity to road access in remote areas. During each electrofishing survey, large sections of river were continuously shocked, and, where available, researchers incorporated all habitat and river morphology (riffle, run, pool). Rainbow trout of a known age (7, 13 and 19 months) from Fraser's Mills Hatchery, grown for stocking in the Bras d'Or Lakes were used as the comparison sample. Fraser's Mills rainbow trout were all-female diploids raised from egg and purchased from Trout Lodge in Washington, USA. Hatchery samples were randomly selected during both 2008 and 2009 to reduce the influence of year to year variation in growth. The sample of unknown origin rainbow trout was captured throughout the Bras d'Or Lakes watershed by anglers using fly, lure and bait.

Electrofished and hatchery sampled fish were anesthetized with a clove oil solution and visually inspected for any marks or abnormalities. Scale samples were taken for analysis from above the lateral line on the left side of the body in the area midway between the dorsal and adipose fins. Fork length, to the nearest millimeter, and weight, to the nearest gram, were measured and recorded. The date and sampling location were also documented. All wild fish were lethally sampled while hatchery samples were released back into Fraser's Mills Hatchery. Samples provided by anglers were processed by anglers and later delivered to NSDFA staff. Data received from anglers included: site location, date, scale samples, length, and weight.

A total of 108 scale samples were analyzed. Known origin samples were comprised of 18 wild rainbow trout (11 fish 2008, 7 fish 2009) and 38 hatchery rainbow trout (11 fish 2008, 27 fish 2009). Fifty-two unknown origin samples were provided by anglers throughout 2008. Scales were mounted on microscope slides labeled with each specimen's corresponding length, weight and relevant capture data. Mounted slides were examined using a compound microscope to select the best quality (non-regenerated, unsoiled) and most representative scale from each fish. Scales were all aged separately by two readers and discrepancies were resolved by re-examining collectively. Based on the stocking date, length and weight, only one fish electrofished could have potentially been of hatchery origin and was, therefore, excluded from the study.

Scale examination revealed all known origin scale samples were less than 2 years of age. Variation in scale growth patterns between hatchery and electrofished samples was also observed and noted. Circuli spacing in hatchery samples appeared more consistent and uniform compared to wild scales. Scales from wild fish displayed fewer circuli spaced closer together in the first year of life (Figure 2).

All selected scales were enlarged using a Canon Microfilm Scanner to 160x magnification (40x microfilm scanner, 4x Xerox machine). Each magnified scale was situated in a comparable position on the Canon Microfilm Scanner and printed on 8.5" x 14" paper. Each individual printed scale was given a unique number and the corresponding length, weight and relevant capture data was transferred to the paper copy. The distance between circuli was measured from the center of one circuli to the center of

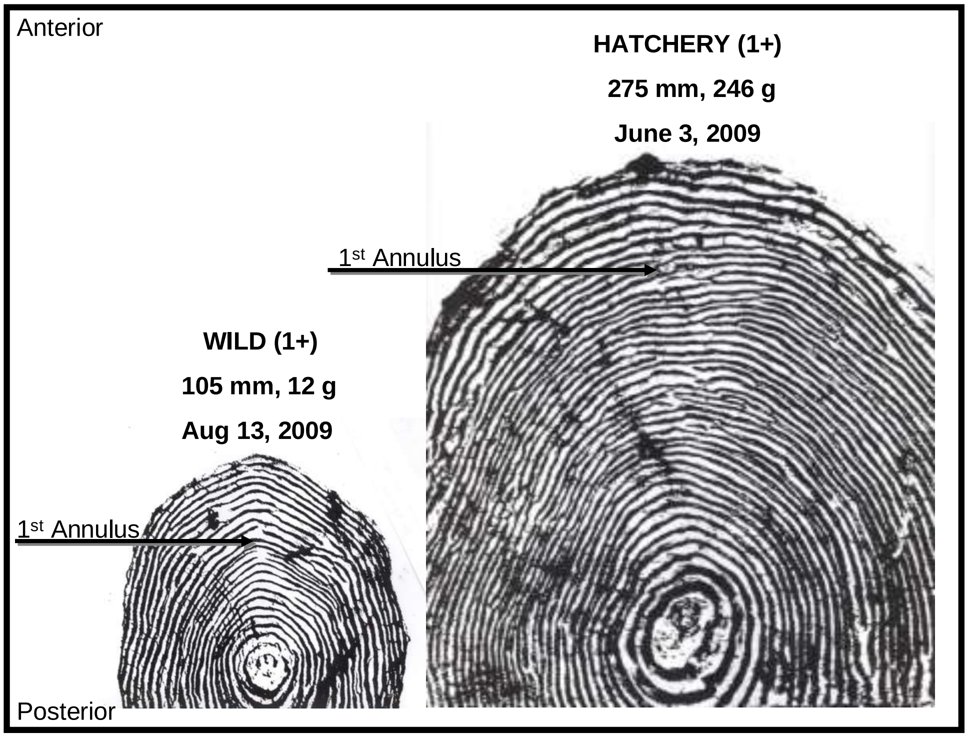


Figure 2. Scale samples from a known 1+ wild and 1+ hatchery rainbow trout depicting growth differences and 1st annulus (year 1 mark). Scales magnified 40x.

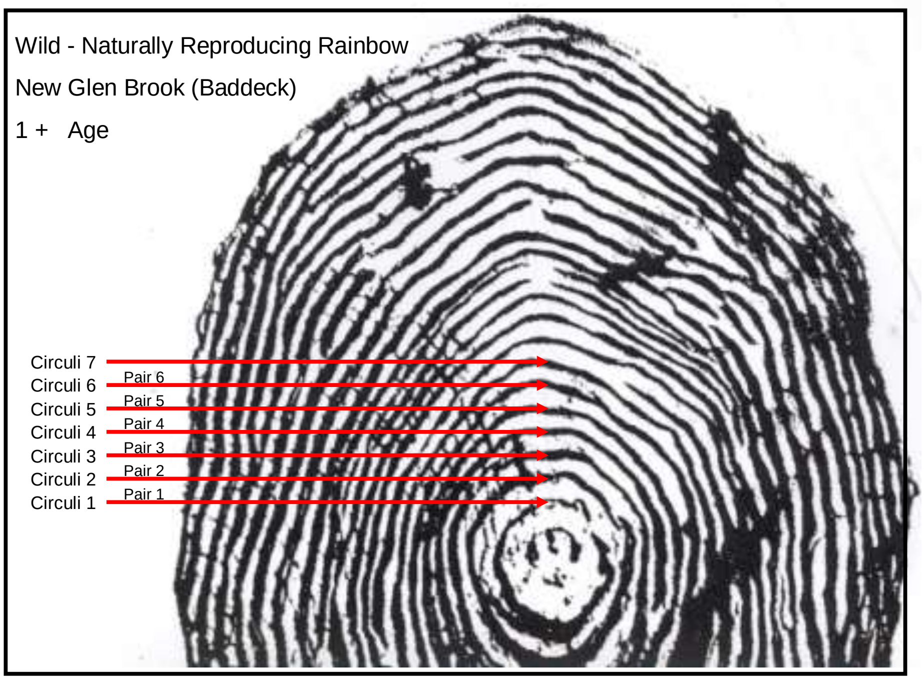


Figure 3. Wild rainbow trout scale depicting circuli pairs used for measurements.

the next circuli, grouped into pairs and recorded to the nearest half millimeter. For example, a circuli pair was the distance between circuli 1-2, measured along the longest axis of the scale. The first circuli surrounding the focus was circuli 1, the first six circuli pairs were measured (Figure 3). All scale measurements were taken by the same person and were measured to the nearest half millimeter. Circuli distances were converted to micrometers (µm) to correspond to the actual distance on each scale using the known magnification.

Because of accelerated growth experienced in hatchery environments, it was hypothesized that the mean width of the first six circuli pairs would be greater in hatchery origin rainbow trout. Therefore, the mean, standard deviation and 95% confidence intervals of the distance between the first six circuli pairs of all wild, hatchery and unknown samples were calculated.

The age of each fish was found for all samples of known origin (hatchery or wild). An exact age was already known for hatchery rainbow trout and the first annulus on wild origin rainbow trout was easily distinguished by the obvious growth patterns displayed on the scales. All wild origin rainbow trout were aged as 1+. During analysis of hatchery scales a false annulus was detected and assumed to have occurred when rainbow trout were introduced to a new environment such as being moved from inside the hatchery to larger, outdoor runs. Hatchery samples displayed many more circuli in the first year of life than the wild samples (Figure 2). Rainbow trout of unknown origin were aged but, due to the presence of false annuli, a definite age could not always be determined.

Circuli pair distances from the different origins were calculated. A Stepwise Logistic Regression was conducted, in Statistical Package for the Social Sciences (SPSS) Version 15, to obtain results on the classification of rainbow trout of unknown origin. The data set consisted of several variables measured for groups of scales from rainbow trout with known origin (hatchery or wild) and a third group with unknown origin. Approximately 70% of the data from the wild and hatchery trout were randomly selected and used to construct a logistic model. The remaining 30% of wild and hatchery data were used for model validation. The dependent variable was wild or hatchery and the predictor variables were weight, length and the six circuli distances. The logistic regression model was used to classify the data with unknown origin as hatchery raised or wild trout. All analyses were performed in SPSS version 15.

RESULTS

Wild juvenile rainbow trout were successfully electrofished in two Bras d'Or Lakes river systems. All wild rainbow trout captured were 1+ years of age. Mean length and weight of wild rainbow trout was 131 mm (2.57, SD) and 29.5 g (18.76, SD) respectively. Mean length and weight of 1+ years old hatchery rainbow trout was 276 mm (2.15, SD) and 252.8 g (58.58, SD) respectively. Hatchery rainbow trout exceeded wild rainbow trout by 145 mm in length and 223.3 g in weight. Hatchery reared samples taken at only 7 months of age, having a mean length and weight of 125 mm (0.74, SD) and 24.8 g (7.08, SD), respectively were similar to 1+ wild rainbow trout. The consistent and intensive feeding regimes of rainbow trout within Nova Scotia hatcheries create rapid growth and measurable differences in scale growth patterns during the first year of life (Figure 2). Length, weight at age and the fact that hatchery stocking occurred after samples were collected, demonstrate that juveniles electrofished from the Bras d'Or Lakes watershed during 2008 and 2009 were of wild origin. Quantitative scale pattern analysis and stepwise logistic regression model analysis further demonstrate that rainbow trout are successfully reproducing in Nova Scotia.

Growth was compared between known origin (wild and hatchery) and unknown origin (angled samples) fish by measuring the distance between circuli (Figure 3). The distance between all six circuli pairs was recorded for each individual fish and the mean distance of all six circuli pairs was calculated for all rainbow trout (Figure 4). The grand mean distance of all six circuli pairs, calculated for all wild, unknown, and hatchery samples was 25.75 µm, 35.03 µm and 38.06 µm, respectively. The grand mean distance of all six circuli pairs of hatchery origin trout is 12.31 µm greater than wild samples.

The total length of six circuli pairs for each fish was determined by calculating the sum of the distance of all six circuli pairs. The mean total length of the six circuli pairs for wild, unknown and hatchery groups was 154.51 µm, 210.16 µm and 228.37 µm, respectively.

The mean distance of the individual circuli pairs (1 - 6) for all wild, unknown (angled) and hatchery samples was calculated. The mean distance of the 6th circuli pairs for all wild, unknown and hatchery samples was 17.53 µm, 28.37 µm and 30.51 µm, respectively. (Figure 5).

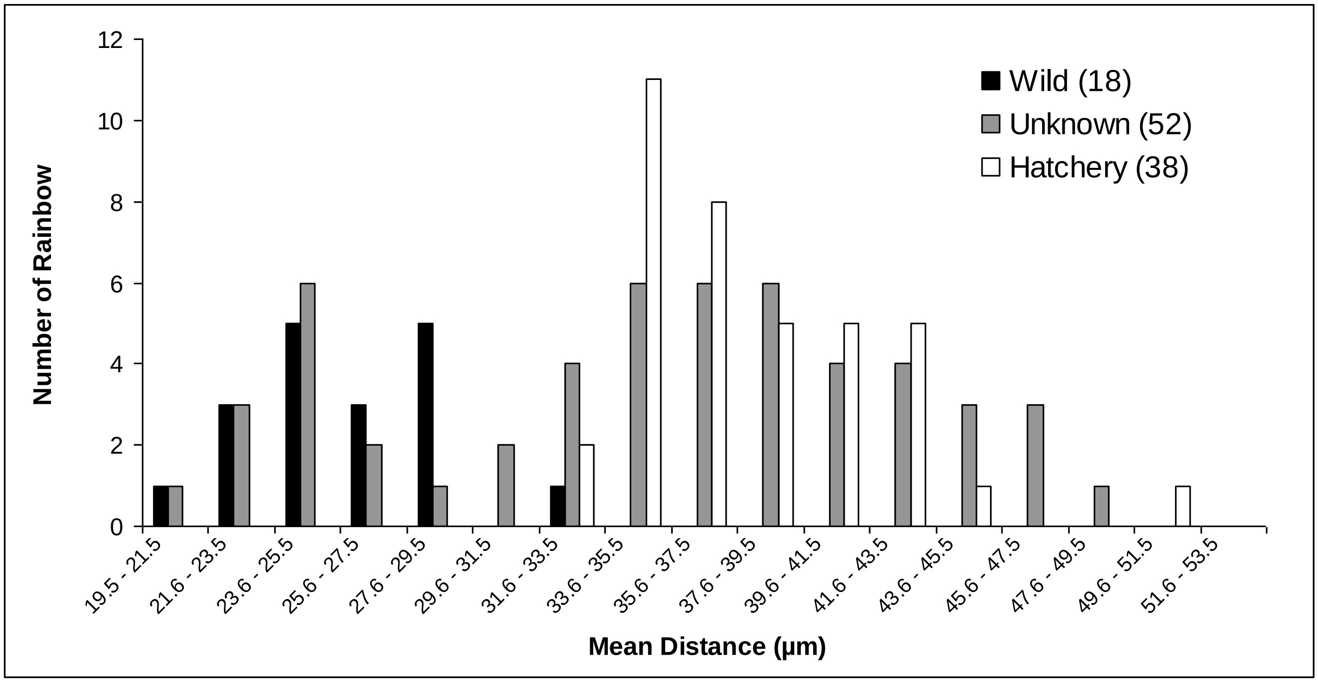


Figure 4. Length frequency distribution of the mean distance of six circuli pairs for each wild, unknown and hatchery rainbow trout.

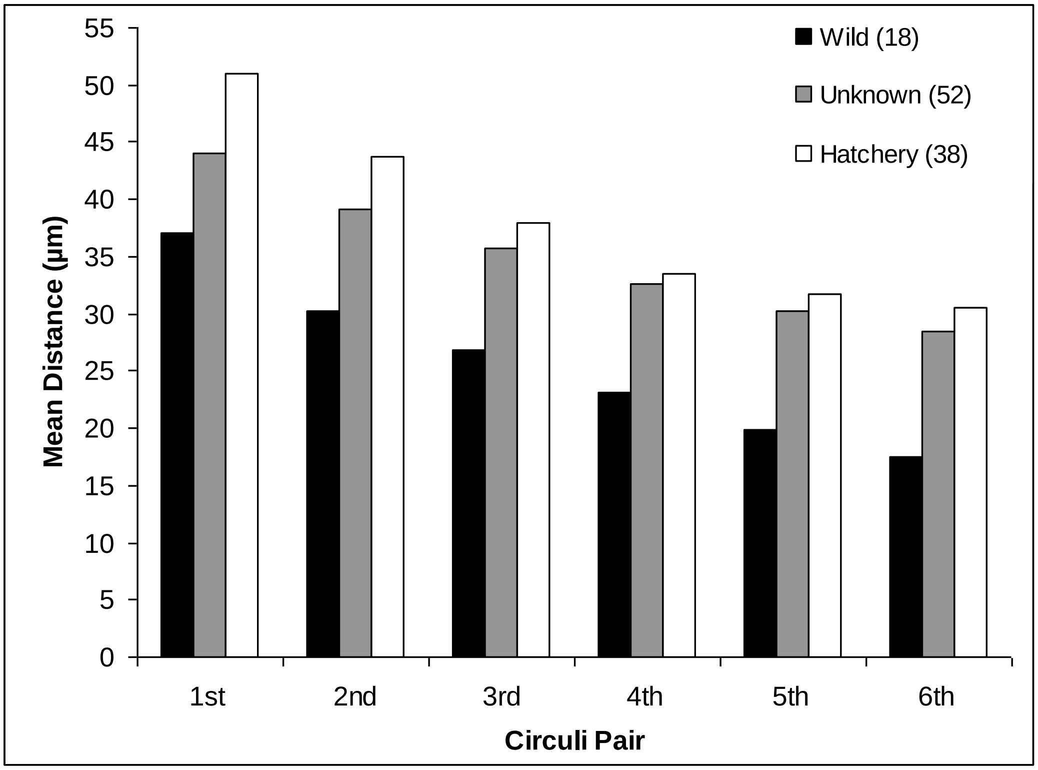


Figure 5. The mean distance of the individual circuli pairs for all wild, unknown and hatchery rainbow trout.

The mean of the circuli pair distances decreases outwards from the focus of the scale towards the anterior edge (Figure 5). Mean distances for the circuli pairs of hatchery rainbow trout were consistently larger than those from the wild. The means for rainbow trout of unknown origin are between those from the hatchery and wild samples (Table 1). Differences between the mean distances of circuli pairs were greater between unknown origin and wild origin fish compared to those between unknown origin and hatchery fish, suggesting that the unknown sample may contain a greater proportion of hatchery fish (Figure 5).

The 95% confidence intervals for the circuli pair distances for the hatchery and the wild rainbow trout do not overlap. Similarly, the 95% confidence intervals for the circuli pairs of trout with unknown origin do not overlap with those of wild origin. There are, however, partial overlaps with the confidence intervals for rainbow trout of unknown and hatchery origin. The only exception is the circuli pair 1 distance (Table 1).

Circuli pair distances in the hatchery raised trout are consistently above the upper quartiles of the wild trout (Figure 6). Thus, the values of circuli pair distances in the range of lower quartile to upper quartile of wild trout do not overlap with those of hatchery raised trout. The corresponding values for those with unknown origin overlap with those from the hatchery and from the wild except in the case of circuli pair 6 for trout with unknown origin and trout with wild origin.

Table 1. The mean (X), standard deviation (SD) and 95% confidence intervals (CI) in µm of the width of the first six circuli pairs of all wild, hatchery and unknown samples of rainbow trout from the Bras d'Or Lakes, Nova Scotia.

	Wild			Hatchery			Unknown		
Circuli Pair	X	SD	CI	X	SD	CI	X	SD	CI
1	36.98	6.36	33.81, 40.14	50.99	8.45	48.21, 53.76	43.99	8.25	41.69, 46.29
2	30.21	5.14	27.65, 32.77	43.67	7.28	41.27, 46.06	39.12	10.23	36.27, 41.97
3	26.91	3.89	24.98, 28.84	37.99	5.22	36.28, 39.71	35.76	9.68	33.06, 38.45
4	23.09	5.27	20.47, 25.71	33.47	4.47	32.00, 34.94	32.57	9.02	30.06, 35.08
5	19.79	5.47	17.07, 22.51	31.74	3.29	30.66, 32.83	30.35	7.41	28.29, 32.41
6	17.53	3.73	15.68, 19.39	30.51	3.66	29.31, 31.71	28.37	8.39	26.03, 30.70

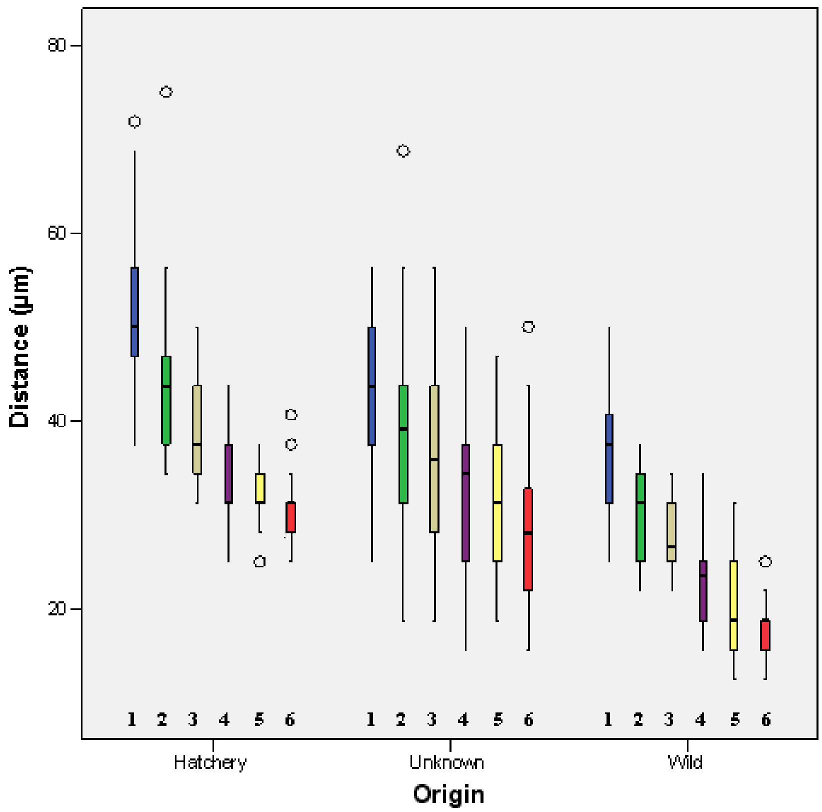


Figure 6. Side-by-side box plots displaying the six circuli pairs (1-6), lower quartile, median, upper quartile and outliers (o) of rainbow trout from different origins (hatchery, unknown and wild).

The sample sizes for the trout from hatchery, unknown and wild origins are 38, 52 and 18, respectively. Stepwise logistic regression analysis using origin (with values wild or hatchery) as the dependent variable and the six circuli pair distances, length and weight of the rainbow trout as predictor variables, showed that only circuli pair 6 distance is a significant predictor. A Hosmer-Lameshow statistic was also calculated indicating a good fit (Sig, 1.000). In the case of selected data with known hatchery or wild origin, the model classified 90.9% of the data from wild samples correctly and 100% of the data from hatchery samples, making the overall percentage correct for selected cases 96.9%. The model classified 100% of all remaining known origin samples used for model validation correctly into hatchery and wild origin.

Out of the 52 rainbow trout with unknown origin, the model predicted 71.15% (± 12.31, 95% c.i.) as hatchery originated. This makes the estimated proportion of wild rainbow trout in the angler catch from the Bras d'Or Lakes watershed 29%.

Discussion

Rainbow trout are successfully reproducing in Nova Scotia and this is the first research to confirm successful reproduction using scale analysis. Quantitative differences in scale characteristics differentiating hatchery stocked and naturally reproducing (wild) rainbow trout suggest that wild rainbow trout represent a substantial proportion of the angled catch from the Bras d'Or Lakes.

Since juvenile rainbow trout grow very rapidly in hatchery conditions and other research (Davis and Light 1985; Stokesbury et al. 2001) illustrates differences in growth through scale analysis in early life stages, scale measurements within the first year of life were used in this study. The first annulus is commonly used in scale analysis to distinguish between hatchery and wild rainbow trout (Davis and Light 1985; Tattam et al. 2002). Misidentification of the first annulus may, however, lead to the inclusion of false annuli when analyzing rainbow trout scales for age and growth. The failure of some salmonids to form their first annulus also reduces the accuracy of this method (Lentsch and Griffith 1987). To eliminate the misidentification of annuli and create a less subjective technique to distinguish the origin of rainbow trout, therefore, the quantitative method used in this study was based on repeatable measurements of circuli spacing. The stepwise logistic regression model used was successful, with a high level of confidence, in accurately differentiating between wild and hatchery rainbow trout and, therefore, can be used as a management tool. All juvenile rainbow trout electrofished from the Bras d'Or Lakes watershed were classified as wild rainbow trout.

Rainbow trout were commercially produced at aquaculture facilities in the Bras d'Or Lakes from 1972 to 2002 and there are currently nine marine rainbow trout aquaculture sites in abeyance (personal communication, Cameron 2010). Large aquaculture escapement events added to the popularity of the fishery and likely contributed to the reproducing population. After these escapements, Hurley Fisheries Consulting (1989) reported runs of rainbow trout in Skye River and, to a lesser extent, in other rivers during the late 1980's.

Stocking of mixed-sex hatchery rainbow trout has occurred in Nova Scotia since the early 1900's. In 2007, however, diploid all-female rainbow trout became the standard for stocking. Hatchery rainbow trout stocked in the Bras d'Or Lakes have not been externally marked. Past stocking and escapement of viable rainbow trout have potentially bolstered spawning biomass. Previously, juvenile rainbow trout have been captured during electrofishing surveys in the Middle River (Robichaud-Leblanc and Amiro 2004), Skye River, Gillis Lake Brook, Blues Brook, and Breac's Brook (Sabean 1983) indicating that reproducing populations may exist. From a management perspective, the successful reproduction of rainbow trout presents several questions. What impacts will this known invasive species have on native fauna? What is the current contribution of wild rainbow trout in the Bras d'Or Lakes watershed? What management decisions need to be implemented to ensure the conservation and protection of native species? How can a sport fishery for rainbow trout be maintained without further impacting native fish species?

Brook trout are Nova Scotia's provincial fish and the most important freshwater species to the recreational fishery. Brook trout are one of the most temperature sensitive salmonids and tend to avoid temperatures greater than 20ºC (Lee and Rinne 1980; Biro 1998; MacMillan et al. 2005). Water bodies in Cape Breton, Nova Scotia offer more cool water refugia than southern sections (MacMillan et al. 2005) of the province and acidification is of little significance in the Bras d'Or drainage basin (Parker et al. 2007); both positive conditions for brook trout to thrive. It is also well documented, however, that brook trout do not successfully compete with other species (Fraser 1978; East and Magnan 1991; Flick and Webster 1992; MacMillan et al. 2005; Halfyard et al. 2008). Rainbow trout, on the other hand, have an optimal water temperature of approximately 19ºC (Cunjak and Green 1986), display a distinct survival advantage over other salmonids at water temperatures above 20ºC (Bear et al. 2007) and compete well with other salmonids (McMichael and Pearsons 1998). Watershed temperature increases associated with global warming, therefore, may create a more suitable environment for rainbow trout, consequently giving them an additional competitive advantage over brook trout. Rainbow trout have shown to be well suited to the Bras d'Or Lakes environment, displaying weight increases of approximately 400% in the first two months of a mark and recapture study (Sabean and Banks 1987).

Overall, although the successful reproduction of the invasive rainbow trout is a management concern in Nova Scotia, it is currently of relatively low priority as recent and past electrofishing surveys demonstrate that limited reproduction has occurred. Although 29% of the harvested (angled) sample was determined to be wild rainbow trout, research has shown that densities of juvenile rainbow trout are far below those of native brook trout and Atlantic salmon (Robichaud-Leblanc and Amiro 2004; MacMillan et al. 2008). Compared to other invasive species such as smallmouth bass and chain pickerel, rainbow trout are believed to have minor influences on native species. Smallmouth bass *Micropterus dolomieui* and chain pickerel *Esox niger* have already had detrimental effects on native fish species by establishing large populations and spreading through trout and salmon waters across the province. Smallmouth bass, well established in southern and central Nova Scotia, were first reported in northern Nova Scotia in 2000 (LeBlanc 2010) after an illegal introduction into Lake Ainslie, the province's largest (5600 ha) natural freshwater lake located just outside the Bras d'Or Lakes watershed (Figure 1). Following the illegal release of this invasive competitor, the catch per unit effort (CPUE) of smallmouth bass in Lake Ainslie increased from 0.3 bass/h in 2003 to 1.63 bass/h in 2008 while brook trout decreased from 1.5 trout/h to 0.11 trout/h during the same time period (LeBlanc 2010). This demonstrates the impacts that invasive species can have on native species, even over short periods of time in large and very productive watersheds.

Currently, triploid rainbow trout are used in Provincial stocking strategies for the Bra d'Or Lakes. The process of creating triploid salmonids is well developed and Dillon et al. (2000) estimate the cost to produce triploid rainbow trout is only 15 percent higher than diploids. Because triploid fish are sterile, they cannot contribute to existing rainbow populations and, therefore, may be a solution to considerably reducing impacts to native stocks while maintaining the sport fishery. Through analysis of creel survey results, Dillon et al. (2000) indicated that triploids provide an angling opportunity equal to that of fertile diploids. Triploid rainbow trout have displayed survival rates 40% - 90% greater than diploid rainbow trout (Teuscher et al. 2003). Compared with all-female and mixed-sex diploid rainbow trout, all-female triploids have higher growth rates (Sheehan et al. 1999). Because triploids put so much energy into growth, an additional beneficial result of using triploid rainbow trout is the possibility of creating a trophy fishery. The current world record for rainbow trout is 21.77 kg (48 lb) (International Game Fish Association 2010). This trophy was angled in Lake Diefenbaker, Saskatchewan, Canada and is widely believed to be a triploid. Stocking only triploid rainbow trout currently seems to be the best management option for the Bras d'Or Lakes watershed as it helps to conserve native species while maintaining this popular sport fishery and costs only marginally more to produce.

The quantitative scale pattern analysis technique used in this study is a nonlethal, relatively inexpensive, repeatable, and, as shown in this research, accurate method to differentiate between wild and hatchery origin rainbow trout. The measurement of distance between the first six circuli pairs eliminates the need to correctly determine annuli and the errors commonly associated with that method. This quantitative scale pattern analysis method will be a valuable management tool for future evaluation of impacts of the use of triploids by hatcheries and aquaculture facilities on the angler catch.

Additional research will need to be conducted to fully assess wild rainbow trout contributions and impacts to the Bras d'Or Lakes watershed. Future studies to determine if juvenile densities are increasing and to assess the impacts of implementing a triploid stocking strategy should be undertaken. All aquaculture and hatchery facilities in Nova Scotia are encouraged to use triploid rainbow trout to reduce impacts associated with successful reproduction on native populations of brook trout and Atlantic salmon.

Acknowledgements

Sincere gratitude is extended to Colin Buhariwalla, Cheryl Wall and Eddie Halfyard for their participation in data collection. Special thanks to Gordie MacKinnon for his participation in this research. The authors also thank the following Nova Scotia Department of Fisheries and Aquaculture staff: Don MacLean, Director, Inland Fisheries Division, for providing resources for this research; Alan McNeill and Jason LeBlanc for editorial comments.

References

Bear, E. A., T. E. McMahon, and A. V. Zale. 2007. Comparative thermal requirements of westslope cutthroat trout and rainbow trout: Implications for species interactions and development of thermal protection standards. Transactions of the American Fisheries Society 136:1113–1121.

Bernard, R. L., and K. W. Myers. 1996. The performance of qualitative scale pattern analysis in the identification of hatchery and wild steelhead (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Sciences. 53: 1727 – 1735.

Bilton, H. T., and G. L. Robins. 1971. Effects of feeding level on circulus formation on scales of young sockeye salmon (*Oncorhynchus nerka*). Journal of the Fisheries Research Board of Canada 28:861-868.

Biro, P. A. 1998. Behavioral thermoregulation during summer by young-of-year brook trout in a lake. Transactions of the American Fisheries Society 127: 212-222.

Boyer, M. C., C. C. Muhlfeld, and F. W. Allendorf. 2008. Rainbow trout (*Oncorhynchus mykiss*) invasion and the spread of hybridization with native westslope cutthroat trout (*Oncorhynchus clarkii lewisi*). Canadian Journal of Fisheries and Aquatic Sciences. 65: 658-669.

Cameron, S. 2010. Aquaculture Licensing Officer, Nova Scotia Department of Fisheries and Aquaculture, Aquaculture Division. Personal communication.

Chadwick, E. M. P., and W. J. Bruce. 1981. Range extension of steelhead trout (*Salmo gairdneri*) in Newfoundland. Naturaliste Canadien, 108, 301-303.

Cunjack, R. A., and J. M. Green. 1983. Habitat utilization by brook charr *(Salvelinus fontina1is)* and rainbow trout *(Salmo gairdneri)* in Newfoundland streams. Canadian Journal of Zoology. 61: 12 14-12 19.

Cunjak, R. A., and J. M. Green. 1986. Influence of water temperature on behavioural interactions between juvenile brook charr, *Salvelinus fontinalis*, and rainbow trout, *Salmo gairdneri*. Canadian Journal of Zoology. 64: 1288-1291.

Davis, N. D., and J. T. Light. 1985. Steelhead age determination techniques. International North Pacific Fisheries Commission document. FRI-UW.8506. Fisheries Research Institute, University of Washington, Seattle, Washington.

Dillon, J. C., D. J. Schill, and D. M. Teuscher. 2000. Relative return to creel of triploid and diploid rainbow trout stocked in eighteen Idaho streams. North American Journal of Fisheries Management 20: 1-9.

East, P. and P. Magnan. 1991. Some factors regulating piscivory of brook trout in lakes of the Laurentian Shield. Canadian Journal of Fisheries and Aquatic Sciences. 48:1735–1743.

Fisheries and Oceans Canada. 2007. Survey of recreational fishing in Canada. Economic Analysis and Statistics, Policy Sector. Full Nova Scotia data set unpublished.

Flick, W. A. and D. W. Webster. 1992. Standing crops of brook trout in Adirondack waters before and after removal of non-trout species. North American Journal of Fisheries Management. 12:783–796.

Fraser, J. M. 1978. The effect of competition with yellow perch on the survival and growth of planted brook trout, splake, and rainbow trout in a small Ontario lake. Transactions of the American Fisheries Society. 107(4): 505-517.

Halfyard, E. A., J. L. MacMillan, and R. J. Madden. 2008. Fecundity and sexual maturity in select Nova Scotia trout populations. Unpublished report. Inland Fisheries Division, Nova Scotia Department of Fisheries and Aquaculture. Pictou, Nova Scotia.

Hurley Fisheries Consulting. 1989. Enhancing the recreational salmonid fishery in the Bras d'Or Lakes: feasibility study. Prepared for Bras d'Or Lakes Recreational Fisheries Ltd. Sydney, Nova Scotia. iii+89p.

International Game Fish Association. 2010. All-tackle records, rainbow trout, viewed 18 July, 2010, <http://www.igfa.org/Records/Fish-Records.aspx?Fish=Trout,%20rainbow&LC=ATR>.

Krauel, D. P. 1976. A summary of the physical oceanography of the Bras d’Or Lake system. The Proceedings of the Bras d’Or Lakes Aquaculture Conference. College of Cape Breton Press. 29-43.

Larson, G. L., and S. E. Moore. 1985. Encroachment of exotic rainbow trout into stream populations of native brook trout in the southern Appalachian mountains. Transactions of the American Fisheries Society 114:195–203.

Larson, G. L., S. E. Moore, and B. Carter. 1995. Ebb and flow of encroachment by nonnative rainbow trout in a small stream in the southern Appalachian mountains. Transactions of the American Fisheries Society 124:613–622.

LeBlanc, J. E. 2010. Geographic distribution of smallmouth bass, *Micropterus dolomieu*, in Nova Scotia: history of early introductions and factors affecting current range. Department of Fisheries and Oceans. Canadian Science Advisory Secretariat Research Document. 2010/028. iv + 25 p.

Lee, R. M. and J. N. Rinne. 1980. Critical thermal maximum of five trout species in the Southwestern United States. Transactions of the American Fisheries Society 109 (6): 632-635.

Lentsch, L. D. and J. S. Griffith. 1987. Lack of first year annuli on scales: frequency of occurrence and predictability in trout of the Western United States. Age and Growth of Fish. Iowa State University Press. 177-188.

MacMillan, J. L., D. Caissie, J. E. LeBlanc and T. J. Crandlemere. 2005. Characterization of summer water temperatures for 312 selected sites in Nova Scotia. Canadian Journal of Fisheries and Aquatic Sciences Technical Report. 2582: 43p.

MacMillan, J. L., D. Caissie, T. J. Marshall, and L. Hinks. 2008. Population indices of brook trout (*Salvelinus fontinalis*), Atlantic salmon (*Salmo salar*), and salmonid competitors in relation to summer water temperature and habitat parameters in 100 streams in Nova Scotia. Canadian Journal of Fisheries and Aquatic Sciences Technical Report. 2819: 41p.

McKenna J. E., and J. H. Johnson. 2005. Juvenile rainbow trout production in New York tributaries of Lake Ontario: implications for Atlantic salmon restoration. North American Journal of Fisheries Management 25:391–403.

McMichael, G. A. and T. N. Pearsons. 1998. Effects of wild juvenile spring chinook salmon on growth and abundance of wild rainbow trout. Transactions of the American Fisheries Society. 127: 261–274.

Metcalf, J. L., M. R. Siegle, and A. P. Martin. 2008. Hybridization dynamics between Colorado's native cutthroat trout and introduced rainbow trout. Oxford Journal of Heredity. 99(2):149–156.

Moore, S. E., B. L. Ridley, and G. L. Larson. 1983. Standing crops of brook trout concurrent with removal of rainbow trout from selected streams in Great Smoky Mountains National Park. North American Journal of Fisheries Management 3:72-80.

Parker, M., M. Westhead, P. Doherty and J. Naug. 2007. Ecosystem overview and assessment report for the Bras d'Or Lakes, Nova Scotia. Canadian Manuscript Report of Fisheries and Aquatic Sciences. 2789: xxii +223 pp.

Robichaud-LeBlanc, K. and P. Amiro. 2004. Assessments of Atlantic salmon stocks in selected rivers of Eastern Cape Breton, SFA 19, to 2003. Canadian Science Advisory Secretariat Research Document. 2004/017. 66p.

Sabean, B. C. 1983. Bras d'Or Lake rainbow trout fishery. Nova Scotia Department of Lands and Forests, Technical Note No. 17. 10p.

Sabean, B. C. and D. B. Banks. 1987. Bras d'Or Lake rainbow trout: growth data. Nova Scotia Department of Lands and Forests, Technical Note No. 47.

Scott, W. B. and E. J. Crossman. 1998. Freshwater Fishes of Canada. Galt House Publications, Oakville, Ontario, Canada. 4: 184-191.

Sheehan, R. J., S. P. Shashteen, A. V. Surresh, A. R. Kapuscinski and J. E. Seeb. 1999. Better growth in all-female diploid and triploid rainbow trout. Transactions of the American Fisheries Society 128: 491-498.

Stokesbury, M. J. W., G. L. Lacroix, E. L. Price, D. Knox, and M. J. Dadswell. 2001. Identification by scale analysis of farmed Atlantic salmon juveniles in Southwestern New Brunswick rivers. Transactions of the American Fisheries Society 130:815-822.

Tattam, I. A., T. A. Whitesel, and Y. Pan. 2003. Scale pattern analysis of selected scale characteristics and the first annulus for distinguishing wild and hatchery steelhead in the Hood River, Oregon. North American Journal of Fisheries Management. 23: 856-868.

Teuscher, D. M., D. J. Schill, D. J. Megargle, and J. C. Dillon. 2003. Relative survival and growth of triploid and diploid rainbow trout in two Idaho reservoirs. North American Journal of Fisheries Management. 23: 983-988.

Thibault, I., L. Bernatchez, and J. J. Dodson. 2009. The contribution of newly established populations to the dynamics of range expansion in a one-dimensional fluvial-estuarine system: rainbow trout (*Oncorhynchus mykiss*) in Eastern Quebec. Diversity and Distributions. 15: 1060-1072

SURVIVAL AND GROWTH RATE DIFFERENCES BETWEEN MARBLE TROUT LIVING IN ALLOPATRY OR IN SYMPATRY WITH RAINBOW TROUT IN UPPER IDRIJCA (SLOVENIA)

S Vincenzi[a], A J Crivellì[b], D Jesensek[c], G Rossi[a], G A De Leo[d]

[a]Dipartimento di Scienze Ambientali, Università degli Studi di Parma, Viale G.P. Usberti 33/A, I-43100 Parma, Italy, simone.vincenzi@nemo.unipr.it, Tel: (+39) 0521 905696

[b]Station Biologique de la Tour du Valat, Le Sambuc,F-13200, Arles, France

[c]Tolmin Angling Association, Modrej 26a, 65216 Most na Soci, Slovenia

[d]Dipartimento di Scienze Ambientali, Università degli Studi di Parma, Viale G.P. Usberti 11/A, I-43100 Parma, Italy

ABSTRACT — We compared two distinct stream sectors where marble trout *Salmo marmoratus* occur in allopatry (MTa) or sympatry (MTs) with rainbow trout (RTs) *Oncorhynchus mykiss* in the Idrijca River (Slovenia) to understand the consequences of the invasion of the nonnative rainbow trout on the native marble trout,. Using data from field surveys from 2002 to 2009, with biannual (June and September) sampling and tagging from June 2004 onwards, we analyzed body growth and survival probabilities of marble trout in each stream sector. The software MARK was used to generate plausible models of apparent survival for rainbow trout and marble trout. Monthly apparent survival probabilities were slightly higher in MTa than in MTs, while RTs showed a lower survival than MTs. Rainbow trout did not depress body growth of sympatric marble trout between sampling intervals. Mean weight of RTs cohorts at age 0+ in September was significantly higher than weight of both MTs and MTa, despite a later emergence. The self-sustaining population of rainbow trout had a stable coexistence with sympatric marble trout, with no clear impact on marble trout body growth and survival probabilities.

INTRODUCTION

The management and control of invasive species is one of the biggest challenges conservation biologists are facing (Allendorf and Lundquist 2003). Among the biological traits facilitating invasions, there is general consensus that r-selected life-history strategy is one of the most important, particularly in the first stages of invasion; an invader with high fecundity, rapid growth, and early maturity often has high potential to establish self-reproducing populations and spread in a novel environment (e.g., Moyle and Marchetti 2006). Marble trout *Salmo marmoratus* is a stream-living salmonid endemic to the Southern Alpine region. Alien rainbow trout *Oncorhynchus mykiss* and brown trout *Salmo trutta* have been introduced in the last 100 years in the distribution area of marble trout. The Adriatic and Danubian basins of Slovenia are among the few locations in Europe where rainbow trout has established self-sustaining populations, despite the continuous stocking of rainbow trout in many European waters for more than 100 years. In this work, we investigated in the field the impact of introduced rainbow trout on a resident marble trout population living in Upper Idrijca (Slovenia).

MATERIAL AND METHODS

Study Area and Species Description

The study took place in the Upper Idrijca area, just below the spring of Idrijca river (Figure 1). The Idrijca River is located in southwestern Slovenia . Description of the biology and ecology of marble trout can be found in Crivelli et al. (2000). Rainbow trout, an Eastern Pacific species (Gall and Crandell 1992), were introduced in the Adriatic basin of Slovenia in the early twentieth century and here established self-sustaining stream-resident populations. Rainbow trout were stocked in Upper Idrijca only once, in 1962, and there rainbow trout established a self-sustaining stream-resident population living in sympatry with genetically pure marble trout. No other fish species live in Upper Idrijca.

Field Data Collection

Sampling surveys were carried out from September 2002 to September 2009 on the whole length of each sector starting from downstream using a gasoline-powered, portable backpack electrofishing unit. Trout were sampled since 2002 in Sector A (marble trout in allopatry, MTa) and in Sector S, where marble trout lives in sympatry (MTs) with rainbow trout (RTs) (Figure 1). In 2002 and 2003 trout were sampled only in September and individuals were not tagged. Trout were individually tagged since June 2004 with biannual sampling, in early June and early September. All captured fish aged ≥1+ were anaesthetized with 2-phenoxyethanol, measured for total length (L_T, to the nearest mm) and weight (W, g) and if sampled for the first time the adipose fin was removed. There was no evident difference in daily water temperature between Sector A and Sector S.

Analysis of Recapture Probabilities and Survival

Capture histories were generated for a fish only if it was sampled at age ≥ 1+, as only trout aged 1+ or older were tagged. Two probabilities can be estimated from a capture history matrix: ϕ, the apparent survival probability, and p, the capture probability (Lebreton et al. 1992). Capture history matrices were used as input files for the software MARK (White and Burnham 1999) to compute maximum likelihood estimates of the apparent survival (ϕ), the recapture probability (p), and their

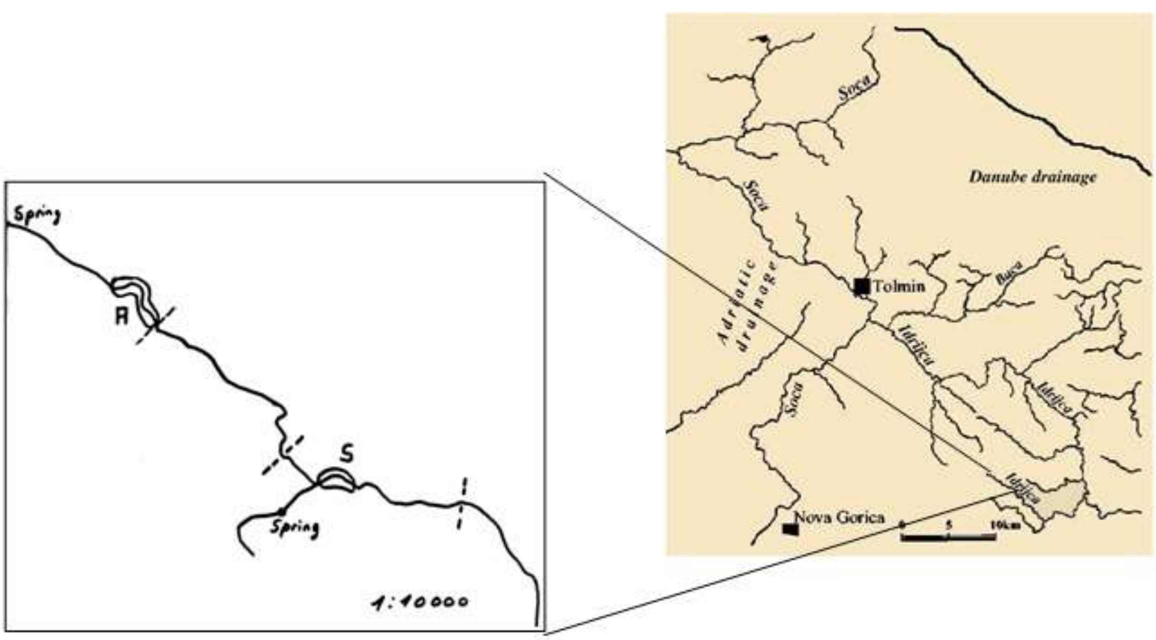


Figure 1. The Upper Idrijca River within the Adriatic basin of Slovenia. Dotted lines represent major waterfalls (height > 10 m). In Sector A, from the waterfall to the spring of the river, marble trout live in allopatry. In Sector S, bounded by two waterfalls, marble trout and rainbow trout live in sympatry.

respective 95% confidence intervals (CI). The Cormack–Jolly–Seber (CJS) model was used as a starting point for the analyses. This model allows both survival and recapture probabilities to vary with time for a single group of animals (Lebreton et al. 1992). The model with the maximum parameterization (global starting model), had for both ϕ and p a multiplicative interaction between year of birth (*yob*), Distribution (*Distr* - MTa, MTs, RTs) and sampling occasion (*t*). The goodness-of-fit (GOF) test of the Cormack–Jolly-Seber (CJS) model was performed using the program U-CARE 2.0 (Choquet et al. 2005). All of the tested survival models were simplified versions of the global starting model. A seasonal effect (*s*) was modeled as a simplification of full time variation, where each year is divided in two periods: the season from June to September (summer), and the season covering the time period between September and June (winter). As length of sampling intervals (summer and winter) was different, we estimated a monthly survival (ϕ) via techniques within the Program MARK (White and Burnham 1999). Model selection among the candidate models was based on the corrected Akaike Information Criterion, AICc. From the global model, recapture probability was modelled first, by allowing the survival probability to vary among *yob*, *Distr* and *t* components. Paired t-tests were used to test for differences in monthly survival from 0+ in September to 1+ in June, between (i) marble trout and rainbow trout living in sympatry, and (ii) marble trout in sympatry and marble trout in allopatry. As fish were not tagged when 0+, we computed monthly survival by simply dividing the estimated number of 0+ in September by the estimated number of 1+ in June and then dividing by nine months. Data for early survival span from September 2004-June 2005 to September 2008-June 2009.

Analysis of Body Growth

Differences in mean ln-transformed weight ($\ln \bar{W}$) of marble trout cohorts at age 0+ and 1+ (in September for 0+ and June for 1+) living in sympatry with rainbow trout and allopatry were tested using t-tests, which were also used to investigate if mean $\ln \bar{W}$ differed between marble and rainbow trout living in sympatry. Mean *ln*-transformed weight of 0+ trout cohorts (marble and rainbow) in September ($\ln \bar{W}_{0+}$) was tested for density dependence using three analysis of covariance (ANCOVA) models. The common covariate in the three models was *ln*-transformed density of the cohort (continuous variable), while the grouping variable was either (i) the combination between Sector and species (MTa, MTs, RTs), (ii) species (MT, RT), (iii) Sector (Sector S, Sector A). We used separated species-specific density of cohorts as predictors. The AIC was used for model selection. Then, we tested with ANCOVAs if body growth between sampling intervals of allopatric and sympatric marble trout born in the same year was different. We chose as response variable the natural logarithm of weight at the end of an interval (ln W_2), while the independent variables were the natural logarithm of weight at the beginning of the interval (ln W_1) and the grouping factor allopatry vs. sympatry. We used only combination of cohorts and years with a minimum of three individuals. For each ANCOVA we tested for the interaction term, that is the heterogeneity in the slopes of lines relating final size to initial size. If the interaction term was not significant, the model was fitted again without the interaction term, thus allowing comparisons of line elevations (i.e. final sizes at a common initial size). We could not include rainbow trout in the ANCOVAs due to limited sample size.

RESULTS

Age, Size and Population Densities

Maximum age was 4+ for sympatric rainbow trout, but very few rainbow trout were older than 1+ -whereas sympatric marble trout reached age 10+ and allopatric marble trout age 8+. Densities in individuals· m^{-2} (for marble and rainbow trout aged 0+ and ≥1+) estimated in September of each year from 2002 to 2009 are shown in Figure 2).

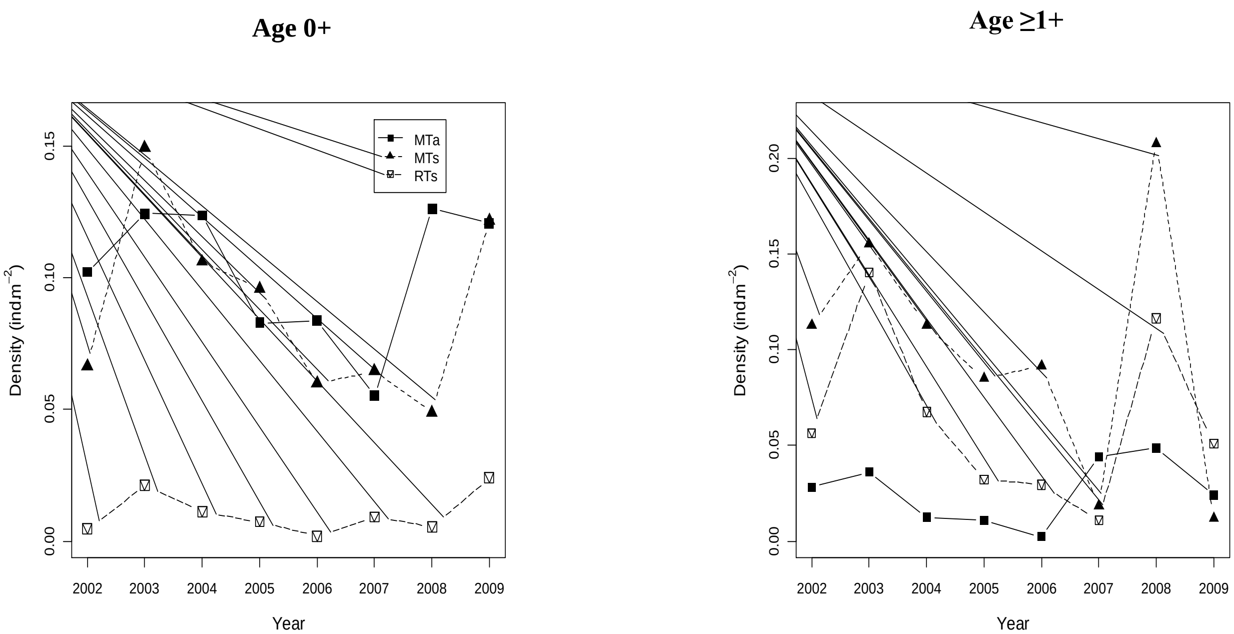


Figure 2. Density of rainbow trout and of marble trout (in individuals m^{-2}) estimated in September from 2002 to 2009 in Sector S (MTs, RTs) and Sector A (MTa). Left, trout aged ≥1+ (mean±sd): MTa, 0.10±0.03 individuals$^{\cdot}$ m^{-2}; MTs, 0.09±0.03; RTs, 0.01±0.08. Right, trout age 0+: MTa, 0.03±0.02; MTs, 0.1±0.06; RTs, 0.07±0.04.

Recapture and Survival Probabilities

Tagged trout were not recaptured outside the sampling sections. The best model of recapture and apparent survival probabilities included only the component Distribution (MTa, MTs, RTs). Models of apparent survival also included season as temporal component, either multiplicative or additive, in addition to the grouping factor Distribution, had very close AICc values (Table 1). Using the best model for inference, apparent survival varied by Distribution, with the highest survival probability for MTa and the lowest for RTs, with a slight overlap of 95% confidence intervals for MTa and MTs (Figure 3). In models including the seasonal component, survival was higher in summer for marble trout in allopatry, and higher in winter for marble trout and rainbow trout living in sympatry (Figure 3). Paired t-tests revealed no significant differences (p>0.05) in monthly September to June survival of juveniles for marble trout in sympatry (mean±sd of survival = 0.93±0.05) *vs* marble trout in allopatry (0.96±0.04). Differences in mean survival between sympatric marble trout and rainbow trout (0.82±0.09) were significant (p<0.05).

Table 1. Model selection for estimation of apparent monthly survival (ϕ) and recapture (p) probabilities for marble trout and rainbow trout living in sympatry and for marble trout living in allopatry in Upper Idrijca. First, we modelled probability of recapture p by keeping the global model of survival. Then, we used the best model for p to model survival probabilities ϕ. For each candidate model we report the AICc and Delta AICc. Only the best three models are presented.

Model		*AICc*	*Delta AICc*
Modelling capture			
ϕ(t*yob*Distr)	p(Distr)	1837.00	0.00
ϕ(t*yob*Distr)	p(sector)	1839.76	2.76
ϕ(t*yob*Distr)	p(s + sector)	1841.49	4.49
Modelling survival			
p(Distr)	ϕ(Distr)	1811.04	0
p(Distr)	ϕ(s*Distr)	1812.64	1.60
p(Distr)	ϕ(s + Distr)	1813.04	1.99

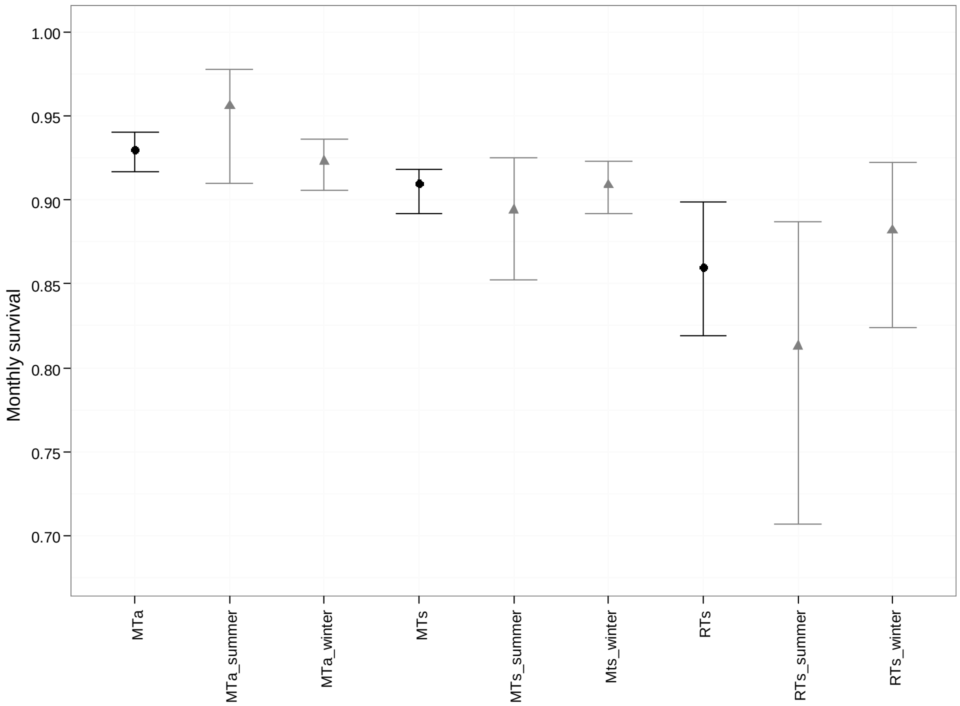


Figure 3. Apparent survival estimates ± 95% CI for the overall best model (circles, black) ϕ (Distr)*p*(Distr) and the best model with a time component (triangles, grey) ϕ (s*Distr)*p*(Distr), where s represents season (summer and winter) and Distr represents trout populations (MTa, MTs and RTs).

Growth and Size Variation

Mean weight of 0+ cohorts of marble trout in September was significantly greater in MTa in 2004, 2005 and 2007 (t-tests, p<0.01) and not significantly different in the remaining years. Mean weight of 0+ cohorts of RTs in September was significant greater than mean weight of 0+ cohorts of MTs in 2004, 2005, 2006 and 2008 (t-tests, p<0.01) and not significantly different in 2007 and 2009. As for marble trout cohorts age 1+ in June, mean weight of MTa was significantly greater in all years (p<0.05) except 2008, when mean weight of MTs was greater. Mean weight of RTs cohorts age 1+ in June was greater (p<0.01) than mean weight of MTs for all years (for 2006 we did not have enough observations).

The model with the best AIC had trout species (MT and RT) as a grouping variable. Natural log of mean weight ($\ln \bar{W}_{0+}$) decreased with increasing cohort density. Relationship between density of cohort age 0+ (D) and mean weight of cohort ($\bar{W}$) was in the form $\ln \bar{W} = \alpha + \beta_1 \ln D + \beta_2 \ln D * TS$, where TS is trout species (RT = 1 or MT = 0) and model estimates were: β_1 = -0.08±0.03; α: RT = 1.02±0.21, MT = 1.21±0.09; β_2=-0.14±0.07 (p<0.01, R2=0.65). The interaction term was significant, thus revealing different slopes for marble trout and rainbow trout.

For 16 combinations of age and cohort we had the necessary sample size for comparison of growth between sampling intervals of MTa and MTs. The interaction term between initial size (ln *W1*) and Sector was never significant in the ANCOVAs. Removal of the interaction term revealed that marble trout cohorts grew equally in allopatry and in sympatry with rainbow trout.

Discussion

The self-reproducing rainbow trout population of Upper Idrijca does not seem to have a noticeable impact on body growth and survival of sympatric marble trout. Densities of marble trout age ≥ 1+ living allopatry or in sympatry with rainbow trout followed a similar pattern over the study period, while density of rainbow trout was noticeably lower. On the other hand, density of young-of-year fish in September was similar in rainbow trout and marble trout living in sympatry, and greater for marble trout living in allopatry in almost all years. Monthly apparent survival probabilities for individuals age≥1+ were slightly lower for marble trout living in sympatry than in allopatry. Apparent survival probabilities of rainbow trout age≥1+ were substantially lower than those of marble trout. September to June survival of age 0+ fish was not significantly different between MTa and MTs, while it was significantly lower for rainbow trout. Greater survival probabilities in the first winter than for subsequent ages were observed for MTa and MTs, while RTs showed a lower survival during the first winter. Several factors, such as overlap of habitat and diet (e.g, Höjesjö et al. 2005), reproductive success (e.g., Thériault et al. 2007), and redd superimposition (e.g., Essington et al. 1998), may be responsible for differences in survival in sympatric salmonids. We did not have information on diet and habitat preferences of rainbow trout living in Upper Idrijca, but the lower survival of rainbow trout appeared to be consistent across life stages. In models of apparent survival including the seasonal component, monthly survival probabilities were higher during summer for MTa, while they were higher in winter for MTs and RTs. Further investigation and more years of data are needed to assess whether the seasonal patterns observed here are consistent and whether the slightly lower survival of sympatric marble trout is caused by competition with rainbow trout.

The density-dependent pattern in growth has been observed in several salmonids species (Grant and Imre 2005). Observed differences in mean weight of MTa and MTs cohorts at age 0+ and 1+ seem to be related to cohort density. It is worth noting that at equal densities, mean weight of 0+ rainbow trout cohorts was higher than that of marble trout cohorts. The multiple ANCOVAs showed that body growth of fish age ≥1+ was similar in sympatric and allopatric marble trout. While it was not possible to use the same approach to compare body growth between sympatric rainbow trout and marble trout living in Sector S, body growth of rainbow trout is remarkably higher than that of both MTa and MTs. Superiority in interference competition is largely determined by body size (e.g., Nakano 1995; Blanchet et al. 2007) and despite a later emergence, rainbow trout clearly showed superior body growth. This trait could facilitate the invasion and establishment of self-sustaining populations of rainbow trout, but on the other hand life-history traits of sympatric marble trout did not seem to be affected by sympatric rainbow trout. In Sector S, marble trout densities exceeded those of rainbow trout (both for fish age 0+ and ≥1+). It might therefore be expected that sympatric marble trout were more growth limited than rainbow trout, and in fact mean weight of rainbow trout at age 0+ in September and 1+ in June was consistently higher than mean weight of marble trout. At the same time, the higher growth rates of rainbow trout did not depress the growth rates of sympatric marble trout and intraspecific mechanisms seems more important in influencing growth rates than interspecific dynamics.

References

Allendorf, F. W., and L. L. Lundquist.2003. Introduction: population biology, evolution, and control of invasive species. Conservation Biology 17:24-30.

Blanchet, S. , G. Loot, G. Grenouillet, and S. Brosse. 2007. Competitive interactions between native and exotic salmonids: a combined field and laboratory demonstration. Ecology of Freshwater Fish 16:133-143.

Choquet, R., A. M. Reboulet, J. D. Lebreton, O. Gimenez,and R. Pradel. 2005. U-Care 2.2 User's Manual, Cefe, Montpellier, France

Crivelli, A., G. Poizat, P. Berrebi, D. Jesensek, and J. F. Rubin. 2000. Conservation biology applied to fish: the example of a project for rehabilitating the marble trout (*Salmo marmoratus*) in Slovenia. Cybium 24:211-230.

Essington, T. E., P. W. Sorensen, and D. G. Paron.1998. High rate of redd superimposition by brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) in a Minnesota stream cannot be explained by habitat availability alone. Canadian Journal of Fisheries and Aquatic Sciences 55:2310-2316.

Gall, G. A. E., and P. A. Crandell. 1992. The rainbow trout. Aquaculture 100:1-10.

Grant, J. W. A., and I. Imre. 2005. Patterns of density-dependent growth in juvenile stream-dwelling salmonids. Journal of Fish Biology 67:100-110.

Höjesjö, J., J. D. Armstrong, and S. W. Griffiths. 2005. Sneaky feeding by salmon in sympatry with dominant brown trout. Animal Behaviour 69:1037-1041.

Lebreton, J. D., K. P. Burnham, J. Clobert, and D. R. Anderson.1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62:67-118.

Moyle, P. B., and M. P. Marchetti. 2006. Predicting invasion success: freshwater fishes in California as a model. BioScience 56:515-524.

Nakano. S. 1995. Competitive interactions for foraging microhabitats in a size-structured interspecific dominance hierarchy of two sympatric stream salmonids in a natural habitat. Canadian Journal of Zoology 73:1845-1854.

Thériault ,V., L. Bernatchez, and J. J. Dodson. 2007. Mating system and individual reproductive success of sympatric anadromous and resident brook charr, *Salvelinus fontinalis*, under natural conditions. Behavioral Ecology and Sociobiology 62:51-65.

White, G. C., and K. P. Burnham.1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:120-139.

TROUT POPULATION RESPONSES TO WHIRLING DISEASE EPIZOOTICS IN MONTANA RIVERS

Thomas E. McMahon[1], James Robison-Cox[2], Jay Rotella[1], Travis Horton[3], and Billie Kerans[1]
[1]Ecology Department, Montana State University, Bozeman, Montana
[2]Mathematical Sciences Department, Montana State University, Bozeman, Montana
[3]Montana Department of Fish, Wildlife, and Parks, Bozeman, Montana

ABSTRACT — Whirling disease has spread rapidly throughout the United States the past 20 years, but predicting its impacts to trout populations has been problematic. Using a database that contained mark-recapture information for 384,938 trout during the years 1980-2007, a before-after control-impact study design was used to analyze data from infected river sections and non-infected reference sections on six blue-ribbon Montana rivers (Missouri, Blackfoot, Bitterroot, Gallatin, Ruby rivers and Rock Creek) having severe whirling disease infections. A Bayesian mark-recapture model indicated that disease had a strong negative effect on abundance of small (200-300 mm total length) rainbow trout *Oncorhynchus mykiss*, with abundance declining an average of 50% (range 30-69%) of pre-disease levels. This marked decline was consistent across all study rivers. In contrast, a parallel decline in larger fish was not observed; instead, the numbers of rainbow trout >300 mm long either remained the same or increased after whirling disease, with the magnitude of the changes varying by river. Rainbow trout of all size classes showed no reduction in growth or condition after a whirling disease outbreak, suggesting that those fish that survive initial infection do not suffer survival or performance deficits even in highly infected systems. As anticipated, there were generally few changes in brown trout density following whirling disease. The exception was in Rock Creek, where decline in rainbow trout from 90% of total trout density to 20-30% after whirling disease was met by a similar magnitude increase in brown trout. Across all rivers, high infectivity levels coincided with low stream flows since 2000, indicating drought may have exacerbated whirling disease impacts on rainbow trout.

INTRODUCTION

Whirling disease, an infection of salmonids caused by the non-indigenous metazoan parasite *Myxobolus cerebralis* has rapidly expanded over the past 20 years (Bartholomew and Reno 2002). The parasite has now been detected in 22 states, and continues to spread, threatening wild salmonid populations (Bartholomew and Reno 2002; Arsan et al. 2007). Whirling disease has led to major declines in high value recreational trout fisheries throughout the western United States (Nehring and Walker 1966; Vincent 1996). In Colorado, an estimated 560 km of premier trout streams have experienced long-term declines in rainbow trout *Oncorhynchus mykiss* populations; in some locales, declines of 90% of rainbow trout density and biomass have persisted for over 10 years (Nehring and Thompson 2003). For example, in the Gunnison River, Colorado, 150-mm and larger rainbow trout during pre-whirling disease years of the 1980s averaged about 3,400 per km, but subsequent population estimates yielded 531 per km in 1998 and 86 in 2003 (Nehring 2006).

Whirling disease was first confirmed in Montana in 1994 following sharp declines in Madison River rainbow trout (Vincent 1996). This discovery precipitated a statewide program to monitor the spread of the parasite using caged sentinel fish (Baldwin et al. 1998). The disease is now found in most western Montana trout rivers at varying severity levels (Vincent 2003).

One of the perplexing problems with whirling disease has been the high variation reported in population responses to infection. Some infected trout populations have severely declined (Vincent 1996; Nehring 2006), whereas others reportedly showed no detectable effects (Modin 1998; Kaeser et al. 2006). In a review of whirling disease impacts in Colorado, such a wide range of responses led Nehring (2006) to conclude that "it is very difficult to predict with any degree of certainty where, when and under what circumstances the impact of *M. cerebralis* might be devastating and where it would be benign." Although the difficulty in forecasting population impacts from the disease is not unexpected given the dynamic and complex nature of the host-parasite-environment relationship (Kerans and Zale 2002), how the parasite affects salmonids at the population level is of key interest for assessing disease impacts; however, there have been few in-depth studies of trout population dynamics following epizootic outbreaks of whirling disease (Karr et al. 2005).

E. R.Vincent (Montana Fish, Wildlife, and Parks, pers. comm.) noted that rainbow trout population declines in some Montana rivers occurred when 50% or more of sentinel fish had disease severity scores of 3 or more on the MacConnell-Baldwin scale (0 = uninfected, 5 = severe infection). Fish with this level of infection exhibit clinical symptoms of disease including whirling behavior, blacktail, skeketal deformities, and poor survival and performance (Ryce et al. 2005; DuBey et al. 2007). However, the linkage between disease severity observed in sentinel fish and disease severity and population effects in wild fish is uncertain. If population-level effects could be tied to a disease-severity threshold measured from sentinel fish, trout population response could be more reliably predicted based on measured infectivity levels in the field, which would thereby result in improved risk-assessment (Bartholomew et al. 2005).

In this study, we used a before-after control-impact (BACI) study design to assess if trout populations in six different river drainages in Montana exhibit similar responses to severe whirling disease epizootics. These drainages have a unique combination of long-term fish population, whirling disease, and environmental data that allow a detailed analysis of population response to a whirling disease epizootic under varying biotic and abiotic factors. We also assessed possible compensatory growth and survival responses to whirling disease outbreaks by examining other metrics in addition to changes in abundance including recruitment, growth, condition, size structure, and trout species composition before and after the onset of whirling disease.

Methods

We estimated trout population change before and after whirling disease outbreaks on the Missouri, Gallatin, Ruby, Blackfoot, and Bitterroot rivers, and Rock Creek. We compared disease 'impacted' sections to un-impacted 'reference' river sections using a before-after (BA) control-impact (CI) design. 'Reference section' was a river reach that had no or low infection (0-2 disease severity ranking) relative to 'impacted' river sections where there has been a sustained infection risk of 50% or more of sentinel cage fish showing moderate to severe lesions (>3 disease severity ranking) indicative of severe infection. Reference sections were located from 16 to 55 km from impacted sections, a sufficient distance such that the sections can be reasonably considered independent of each other. Infection severities were based on sentinel fish cages deployed on each river since the mid-to-late 1990s (Vincent 2003).

Trout population data were obtained from Montana Fish, Wildlife, and Parks records. Trout population data have been collected in a consistent fashion on Montana rivers since the early 1980s using electrofishing mark-recapture techniques over the same, long sampling sections (1.5-9.0 km) at multiple sites within each drainage (Vincent 1982). The final database was comprised of 384,938 trout collected during the years of 1980 to 2007. Population estimates were generated for all length classes of rainbow trout and brown trout *Salmo trutta* using a Bayesian mark-recapture population estimator developed specifically for this study. Bayesian estimators offer advantages over traditional maximum likelihood estimators (Link and Barker 2010), particularly when the number of parameters is large, as it was for these data, when some of the parameters are related in a hierarchical manner, and when there is interest in estimating additional quantities that are derived from the estimated parameters. The year of first severe whirling disease (WD) infection for the 'infected' reach demarcated the 'before' versus 'after' population estimates. For each site, averaged population size before that time point was used to create a baseline 'before whirling disease' abundance estimate. To quantify population changes after whirling disease, all 'after whirling disease' abundance estimates were divided by their respective baselines, and the population change values were transformed using the $\log_2$ scale because on this scale a doubling of population will be conveniently indicated as a +1 and a halving by -1. Three models were fitted for each set of data: (Model 1) no whirling disease effect (no BA x CI interaction); (Model 2) whirling disease effect (BA x CI interaction term included); (Model 3) whirling disease effect by river (additional variable added to account for variation in response to the disease among individual rivers). Model outputs were generated for 2,400 runs using WinBUGS, a Bayesian software program for conducting Markov chain Monte Carlo analyses (Lunn et al. 2000). We used the 'deviance information criterion' or DIC, an information-theoretic based model selection procedure that can be used in a Bayesian setting (Spiegelhalter et al. (2002) to select among the three competing models. The BACI analyses were run for the four rivers with adequate BACI data: Blackfoot, Gallatin, Missouri, and Ruby rivers; all four rivers had both reference and severely infected sections and multiple years of post-whirling disease data. Two other study rivers, Rock Creek and the Bitterroot River, were treated separately; Rock Creek had before-after data but no reference section, and the Bitterroot River had only one year post-whirling disease data. In addition to the BACI analysis comparing population responses by individual length classes, we also compared total density, biomass, relative species composition, relative weight *Wr*, and growth rate.

RESULTS

Model analysis revealed a strong negative effect of whirling disease on abundance of small rainbow trout (model 2 favored as best model). For all rivers combined, the median proportional decline in small rainbow trout 200-300 mm long after whirling disease was greater than two-fold (0.55 or -1.15 on the $\log_2$ scale; Figure 1). For individual rivers, the decline was highest for the Gallatin (0.69) and Missouri rivers (0.59) and less so for the Blackfoot (0.46) and Ruby (0.30) rivers. In contrast, larger rainbow trout (300-450 mm) showed a significant positive increase (1-2 fold increase) after whirling disease, depending on river (model 3 favored). The strongest increase was shown in the Gallatin and Missouri rivers, and little to no response in the Ruby and Blackfoot rivers. For the largest rainbow trout size class (475-500 mm), all three models had nearly equal DIC values, indicating all three were equally plausible, suggesting the lack of a strong effect of whirling disease.

The before-after analysis of Rock Creek, which lacked a disease-free reference section, showed an even stronger negative whirling disease effect. For rainbow trout 200-300 mm long, there was nearly a four-fold (-1.99 on $\log_2$ scale) decline in abundance after whirling disease (Figure 2). As in other rivers, there was a significant increase in the largest rainbow trout (425-450 mm) after whirling disease, although this response was observed in only one of the two sampling sections. There was only one year of post-whirling disease population data for the main stem Bitterroot River. However, it is noteworthy that the abundance of small rainbow trout 200-300 mm in the whirling disease-positive Darby section was only 30 fish per km in the post-WD year of 2005, 90% below the long-term pre-whirling disease average density of 286 per km from 1989-2002. In contrast, the abundance of small rainbow trout in the reference section (Bell) was similar across all years.

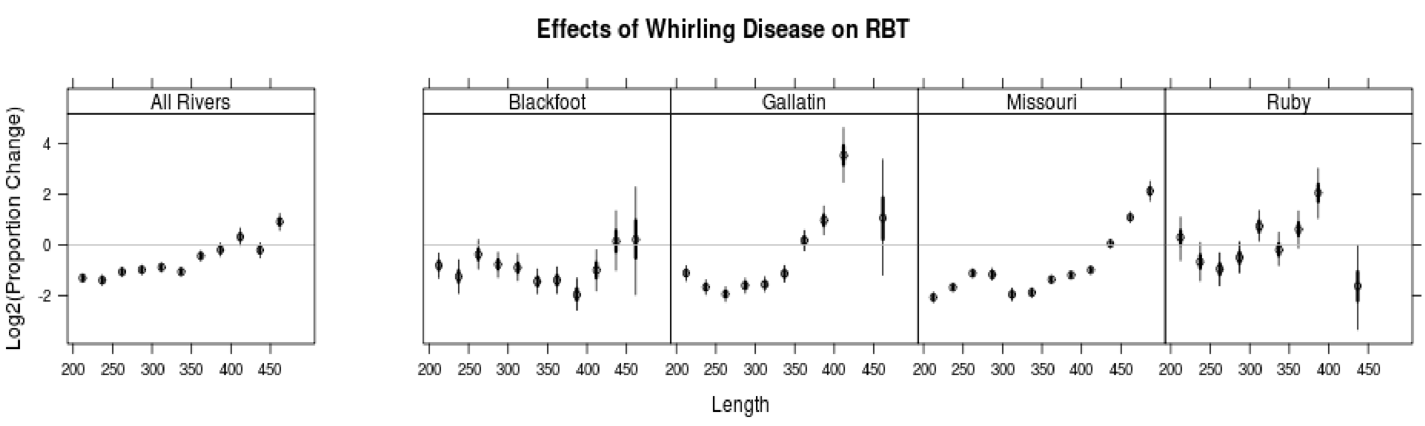


Figure 1. Proportional change in rainbow trout (RBT) density (by length class) after whirling disease outbreaks in four Montana rivers. Analyses based on data collected from 1980 to 2007. The "0" line indicates no difference between reference and whirling disease-infected sites. Values below the line indicate a decrease in density, and values above the line indicate an increase in density compared to the baseline 'pre-whirling disease' density.

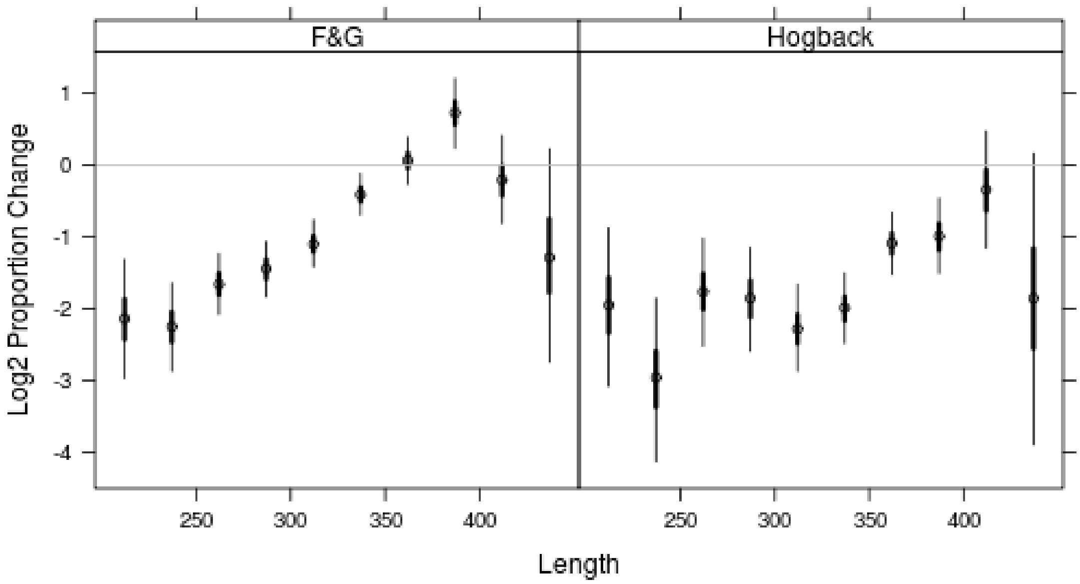


Figure 2. Proportional change in rainbow trout density by length class in two river sections of Rock Creek, Montana, after whirling disease (F&G=Fish and Game section). Analyses based on data collected from 1981 to 2006.

As expected, whirling disease had little effect on brown trout abundance of any size class (model 1 most favored; Figure 3). Rock Creek was an exception, with significant positive increases in abundance of some size classes after whirling disease. The strongest response was shown for brown trout 325-400 mm long, where abundance increased about 2-to-4-fold after WD.

Rainbow trout were the predominant trout in all study rivers, ranging in proportion from 60-90% pre-whirling disease. Most rivers showed little change in the overall proportion of rainbow trout and brown trout pre- and post-whirling disease, with the exception of Rock Creek, which showed a major shift from rainbow trout to brown trout over the study period, from >90% rainbow trout to about 20-30% of total density from 2000-2006.

Whirling disease did not appear to adversely affect trout condition. Trout in the Missouri River were the only group that showed a significant decline in *Wr* after whirling disease. However, this decline was observed in rainbow trout in both the disease-positive (Craig) and disease-negative (Cascade) sections so that the increase could not be clearly attributed as a response to whirling disease. Whirling disease appeared to result in increased

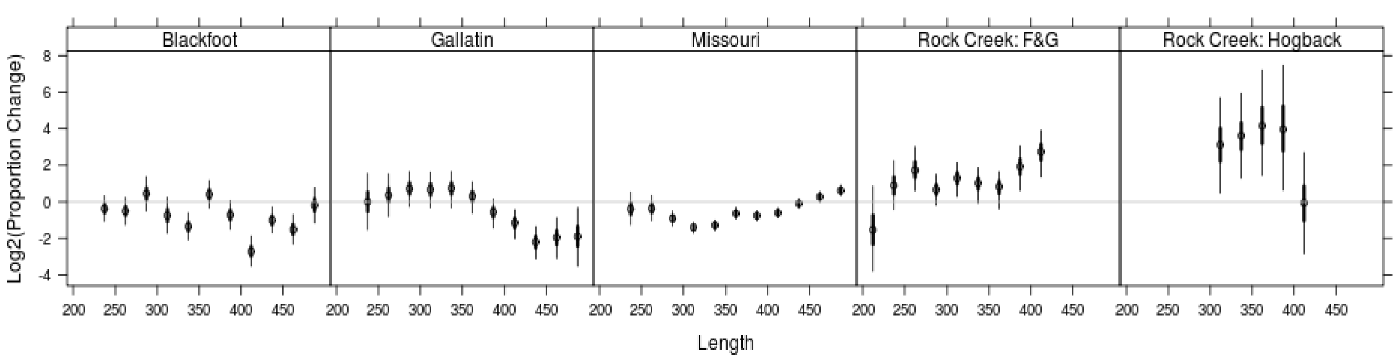


Figure 3. Proportional change in brown trout density in four Montana rivers after whirling disease. Analyses based on data collected from 1980 to 2007.

growth of larger rainbow trout and brown trout. Mean length at age-4+ rainbow trout in the disease-positive Craig section was 499 mm after whirling disease, an increase of 31 mm from pre-whirling disease years (P = 0.03). Mean length of age-3 rainbow trout was also substantially higher after whirling disease (+22 mm; P = 0.07). In contrast, there were no differences in length at age for any age class among rainbow trout from the Cascade reference section. Age estimates were not available for any other river, limiting comparisons of survival and growth among specific age groups.

Discussion

Overall, we found that whirling disease outbreaks led, on average, to a 50% decline (range 30 to 69%) in rainbow trout in the 200-300 mm size class. The decline is likely due to poor survival of age-0 cohorts, because rainbow trout < 9 weeks of age and <40 mm in length are the most highly vulnerable life stage to whirling disease (Ryce et al. 2005). Contrary to our expectations, the marked decline in smaller rainbow trout after whirling disease outbreak did not lead to major declines in larger fish, as observed in Colorado rivers (Nehring and Thompson 2003). Instead, the numbers of rainbow trout >300 mm long either remained the same or increased after whirling disease, with the effects varying by river. Though the lack of decline could be partially attributed to a lag effect wherein low age-0 recruitment has not yet had time to negatively affect subsequent numbers of larger fish, rivers with 5 or more years since the inception of high infectivity (Blackfoot, Gallatin, and Missouri rivers, Rock Creek) showed a similar pattern of stable or increasing numbers of larger trout. The rapidity of the response in abundance, particularly among very large rainbows >400 mm long, coupled with increased growth shown by large Missouri River rainbow trout following whirling disease, suggests a compensatory response in survival and growth. Small and large fluvial rainbow trout share a similar (insectivorous) diet and display strong inter-cohort competition; marked reductions in the density of small trout, as observed in our study, has been shown to increase growth and survival of large trout (Nordwall et al. 2001). In the Missouri River, adult declines have been anticipated for many years once older rainbow trout died out, but the adult population remains robust and the average size of large trout continues to increase, suggesting that survival and growth of large, old rainbow trout in river systems may be much more flexible than previously thought. More detailed examination of age, growth, and survival of these older fish would be a fruitful area for further research.

Why the recruitment declines in rainbow trout we observed were not as severe as those in previous reports of trout response to whirling disease epizootics is uncertain. Declines of juvenile trout recruitment by 90% or more were reported for multiple Colorado rivers (Nehring and Thompson 2003; Nehring 2006) and for the Madison River in the first 8 years following whirling disease outbreak (Vincent pers. comm.) compared to the 50% decline observed in our study. High infectivity of young trout by the parasite requires high spatial overlap between infective spore release and fry emergence within a relatively narrow time window (Downing et al. 2002; MacConnell and Vincent 2002). We suspect that the lack of severe recruitment decline of juvenile rainbow trout was due to the continued presence of uninfected spawning areas even in highly infected rivers (Granath et al. 2007; Pierce et al. 2009). Recruitment from these sites likely serves to maintain rainbow trout populations in these systems, although the relation between recruitment sources and infection risk has not been investigated in detail (but see Pierce et al. 2009).

As expected, whirling disease did not appreciably influence brown trout abundance. Brown trout populations remain largely unchanged in multiple Colorado rivers experiencing major declines in rainbow trout after whirling disease outbreaks (Nehring 2006). In our study, reductions in rainbow trout biomass or density after whirling disease were generally compensated to a similar degree by an increase in biomass and density of brown trout. Though the two species generally occupy different fluvial habitats as adults, the two species may still compete for preferred habitats (Gatz et al. 1987). The major shift in dominance from rainbow trout to brown trout observed in Rock Creek has not been reported in any other rivers following a whirling outbreak. Berg (2004) hypothesized that a combination of high infection risk, low flows, and warmer temperatures over the past 10 years has likely promoted this shift to brown trout dominance in that system.

High infectivity and recruitment declines in juvenile rainbow trout in our study occurred concurrently with significant drought during 2000-2007. Summer flows during this period were 25% or more below the long-term average flow in at least 6 of the last 7 years. Many possible measures of flow were considered as possible covariates in the BACI model, and some of them improved the model fit, suggesting an interaction of whirling disease and flow, with lower flows and high whirling disease both negatively affecting young rainbow trout in the years since 2000.

The association between flow and rainbow trout population response to whirling disease lends support to the hypothesis that lower flows contribute to higher infectivity of salmonid hosts by *M. cerebralis* (MacConnell and Vincent 2002; Hallett and Bartholomew 2008). The reduced velocities at lower flows are thought to promote retention and accumulation of infective stages, settlement of salmonid carcasses, and the deposition of fine sediments that create habitat and a food source for the tubificid worm host (Kerans and Zale 2002; Hallett and Bartholomew 2008).

Conclusions and Management Considerations

Outbreaks of whirling disease epizootics in our Montana study rivers led to an average 50% decline in juvenile rainbow trout. Although the degree of recruitment decline in relation to infection grade level was not evaluated in this study, the results suggest that marked recruitment declines will occur in wild rainbow trout populations when sentinel cage infection levels exceed 50% or more with grade $\geq$3.

The marked decline in small rainbow trout after whirling disease outbreaks did not always lead to reduction in abundance of medium to large rainbow trout. Why severe recruitment declines in larger trout were not observed is uncertain. It is hypothesized that the continued presence of uninfected spawning and early rearing areas within otherwise highly infected rivers, may be the primary mechanism that buffers recruitment, although increased resistance to whirling disease (Miller and Vincent 2009) is another possible explanation.

Increased growth and survival of adult rainbow trout in the face of a whirling disease epizootic may be the result of a compensatory response to marked reductions in density of small rainbow trout. Continued long term monitoring of survival and growth of various size and age classes is needed to determine the stability of this pattern or if maintenance of adult recruitment is only a transitory response.

The lack of decline in growth or condition of rainbow trout after a whirling disease outbreak suggests that young fish that do survive the infection window of high susceptibility do not suffer later survival or performance deficits even in highly infected systems. Recruitment from whirling disease-free spawning and early rearing areas appears crucial for preventing collapse of rainbow trout populations. Protection and enhancement of a diversity of spawning areas and spawning and rearing life histories appears to allow resilience in the face of high infectivity (Pierce et al. 2009). More research is needed to test the proposal that habitat improvement of key infected spawning areas can reduce infectivity and result in population rebound (Thompson and Nehring 2004).

Continued monitoring of whirling disease severity in Montana rivers is needed to further build on the long term database used in this study, as our data suggest that population-level effects have not fully stabilized. Moreover, we found the extent of disease monitoring was limited in many rivers, making it difficult to adequately measure disease severity changes over time.

Given that some cutthroat trout subspecies (DuBey et al. 2007) and mountain whitefish *Prosopium williamsoni* (MacConnell et al. 2000) show even more susceptibility to whirling disease than rainbow trout, population declines of a similar or greater magnitude would also be expected to occur in native cutthroat trout *Oncorhynchus clarkii* and mountain whitefish populations if this infectivity threshold is exceeded.

Drought-driven lower flows may increase whirling disease infectivity. Anticipated declines in summer low flows from climate change will likely lead to further declines of whirling disease-susceptible trout species such as rainbow trout and cutthroat trout and replacement by more resistant species such as brown trout. Research is needed to test ideas to reduce increases in anticipated whirling disease severity in the face of climate change via flow manipulation (Hallett and Bartholomew 2008) and riparian vegetation and channel restoration measures that control temperature increases, maintain stream recharge, and reduce tubificid worm fine sediment habitat (Pierce et al. 2009).

Acknowledgements

The National Partnership on Management of Wild and Native Coldwater Fisheries provided funding for the study. We thank the many FWP biologists who graciously shared their knowledge and population data. The data represent the collective effort of many individuals over many years, and we acknowledge the dedication required to perform the electrofishing estimates under often-adverse conditions. Dick Vincent offered guidance during study initiation and helped identify potential study rivers. He was instrumental in developing a standardized protocol for electrofishing Montana's trout rivers in the 1970s, and for the past 15 years guided the collection of whirling disease infection data in the state, all of which made our study possible.

References

Arsan, E.L., S.D. Atkinson, S.L. Hallett, T. Meyers, and J.L. Bartholomew. 2007. Expanded geographical distribution of *Myxobolus cerebralis*: first detections fro Alaska. Journal of Fish Diseases 30:483-491.

Baldwin, T.J., J.E. Peterson, G.C. McGree, K.D. Staigmiller, E.S. Motteram, C.C. Downs, and D.R. Stanek. 1998. Distribution of *Myxobolus cerebralis* in salmonid fishes in Montana. Journal of Aquatic Animal Health 10: 361-371.

Bartholomew, J.L., and P.W. Reno. 2002. The history and dissemination of whirling disease. Pages 3–24 *in* J. L. Bartholomew and C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland.

Bartholomew, J.L., B.L. Kerans, R.P. Hedrick, S. McDiarmid and J.R. Winton. 2005. A risk assessment based approach for the management of whirling disease. Reviews in Fisheries Science 13: 205-230.

Berg, R.K. 2004. Historical and present day perspectives on fish populations in the Rock Creek drainage in west central Montana, 1971-2004. Montana Fish, Wildlife, and Parks Federal Aid Report F-113-R1/F-113-R2, Missoula.

DuBey, R.J., C.A Caldwell, and W.R. Gould. 2007. Relative susceptibility and effects on performance of Rio Grande cutthroat trout and rainbow trout challenged with *Myxobolus cerebralis*. Transactions of the American Fisheries Society 136:1406-1414.

Downing, D.C., T.E. McMahon, B.L. Kerans, and E.R. Vincent. 2002. Relation of spawning and rearing life history of rainbow trout and susceptibility to *Myxobolus cerebralis* infection in the Madison River, Montana. Journal of Aquatic Animal Health 14:191-203.

Gatz, A.J.,Jr., M.J. Sale, and J.M. Loar. 1987. Habitat shifts in rainbow trout: competitive influences of brown trout. Oecologia 74:7-19.

Granath, W.O., Jr., M.A. Gilbert, E.J. Wyatt-Pescador, and E.R. Vincent. 2007. Epizootiology of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease in the Rock Creek drainage of west-central Montana. Journal of Parasitology 93:104-119.

Hallett, S.L., and J.L. Bartholomew. 2008. Effects of water flow on the infection dynamics of *Myxobolus cerebralis*. Parasitology 135:371-384.

Kaeser, A.J., C. Rasmussen, and W.E. Sharpe. 2006. An examination of environmental factors associated with *Myxobolus cerebralis* infection of wild trout in Pennsylvania. Journal of Aquatic Animal Health 18:90-100.

Karr, J.R., S.B. Adams, and L.S. Fore. 2005. Recommendations for data management, synthesis of knowledge, and future research under the Whirling Disease Initiative. Final report to Whirling Disease Intiative Advisory Panel. Montana Water Center, Bozeman.

Kerans, B.L., and A.V. Zale. 2002. The ecology of *Myxobolus cerebralis*. Pages 145-166 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society, Symposium 29, Bethesda, Maryland, USA.

Link, W.A., and R.J. Barker. 2010. Bayesian inference with ecological applications. Academic Press, New York. 339 pp.

Lunn, D.J., A. Thomas, N. Best, and D. Spiegelhalter. 2000 WinBUGS -- a Bayesian modelling framework: concepts, structure, and extensibility. Statistics and Computing 10:325-337.

MacConnell, E. and E.R. Vincent. 2002. The effects of *Myxobolus cerebralis* on the salmonid host. Pages 95-107 *in* J.L. Bartholomew and J. C. Wilson, editors. Whirling disease: reviews and current topics. American Fisheries Society Symposium 29, Bethesda, Maryland.

MacConnell, E., A.V. Zale, and M. A. Quinn. 2000. Susceptibility of mountain whitefish to *Myxobolus cerebralis*. Pages 107-108 *in* Proceedings of the

6th Annual Whirling Disease Symposium. Whirling Disease Foundation, Bozeman.

Modin, J.G. 1998. Whirling disease in California: a review of its history, distribution, and impacts, 1965-1997. Journal of Aquatic Animal Health 10:132-142.

Nehring, R.B. 2006. Colorado's cold water fisheries: whirling disease case histories and insights for risk management. Special Report Number 79. Colorado Division of Wildlife.

Nehring, R.B., and P.G. Walker. 1996. Whirling disease in the wild: the new reality in the Intermountain West. Fisheries 21:28-32.

Nehring, R.B., and K.G. Thompson. 2003. Whirling disease investigations. Colorado Division of Wildlife, Final Report, Federal Aid Project F-237-R-10. Fort Collins.

Nordwall, F., I. Naslund, and E. Degerman. 2001. Intercohort competition effects on survival, movement, and growth of brown trout (*Salmo trutta*) in Swedish streams. Canadian Journal of Fisheries and Aquatic Sciences 58:2298-2308.

Pierce, R., C. Podner, M. Davidson, and E.R. Vincent. 2009. Correlation of fluvial rainbow trout spawning life history with severity of infection by *Myxobolus cerebralis* in the Blackfoot River basin, Montana. Transactions of the American Fisheries Society 138:251-263.

Ryce, E.K., A.V. Zale, E. MacConnell, and M. Nelson. 2005. Effects of fish age versus size on the development of whirling disease in rainbow trout. Diseases of Aquatic Organisms 63:69-76.

Spiegelhalter, D.J., N.G. Best, B.P. Carlin, and A. van der Linde. 2002. Bayesian measures of model complexity and fit. Journal of the Royal Statistical Society Series B (Statistical Methodology): 64:583-639.

Thompson, K.G., and R.B. Nehring. 2004. Evaluating the efficacy of physical habitat modification to reduce the impacts of *Myxobolus cerebralis* in streams. Pages 49-50 *in* Proceedings of the Tenth Whirling Disease Symposium, Salt Lake City, UT.

Vincent, E.R. 1982. Fish population estimate techniques in Montana. Proceedings of the Annual Conference Western Association of Fish and Wildlife Agencies 62:109-115.

Vincent, E.R. 1996. Whirling disease and wild trout: the Montana experience. Fisheries 21(6):32-33.

Vincent, E.R. 2003. Whirling disease study update-2002, Bitterroot River drainage sentinel cage studies. Montana Fish, Wildlife, and Parks report, Bozeman.

WHIRLING DISEASE IN THE UNITED STATES: A SUMMARY OF PROGRESS IN RESEARCH AND MANAGEMENT

Leah C. Steinbach Elwell

Program Director, Center for Aquatic Nuisance Species, P.O. Box 1429, Livingston, MT 59047 (406) 222-7270 **leah@stopans.org**

Kajsa Eagle Stromberg

Watershed Coordinator, Idaho Dept. Environmental Quality, 2110 Ironwood Pkwy, Coeur d'Alene, ID 83814 (208) 666-4633 **Kajsa.Stromberg@deq.idaho.gov**

Eileen K.N. Ryce

NS Coordinator, Montana Fish, Wildlife & Parks, 1420 East 6th Ave., Helena, MT 59620 (406) 444-2448 **eryce@mt.gov**

Jerri L. Bartholomew

Associate Professor Oregon State University 524 Nash Hall, Corvallis, OR, 97331 (541) 737.1856 **Jerri.Bartholomew@oregonstate.edu**

ABSTRACT — Whirling disease is known in the United States as a serious fish health issue associated with economic and ecological impacts including salmonid population effects. Whirling disease is caused by the microscopic parasite *Myxobolus cerebralis*, whose complicated life cycle relies upon two aquatic hosts: an oligochaete worm *Tubifex tubifex* and a salmonid fish. Understanding whirling disease requires comprehensive knowledge of the parasite, its hosts, and the ecology of the aquatic environment.

Myxobolus cerebralis is detected in new locations each year and is likely to persist in North America. Thus, the key for future management efforts lies in predicting and managing the risks associated with whirling disease. Fortunately, extensive research is available to provide information about whirling disease and a broad suite of management tools and techniques can be employed to prevent the spread of *M. cerebralis* and to mitigate its impacts. In 2009, a review of current research and management was completed to summarize current knowledge on whirling disease and *M. cerebralis* (Elwell et al. 2009). It built on earlier reviews, focusing on new research into invertebrate host ecology, epidemiology within salmonids, management efforts, and improved diagnostic techniques. This article summarizes key points including risk factors and impacts of the parasite.

INTRODUCTION

Myxobolus cerebralis is a parasite species first described in Europe more than 100 years ago and introduced to North America more than 50 years ago. Whirling disease is caused by the microscopic parasite *Myxobolus cerebralis*, whose complicated life cycle relies upon two aquatic hosts: an oligochaete worm *Tubifex tubifex* and a salmonid fish. The parasite also has two important stages; the myxospore and the triactinomyxon infective to *T. tubifex* and salmonids respectively. Understanding whirling disease requires comprehensive knowledge of the parasite as well as the ecology of the aquatic environment, invertebrate host, and the fish host. Whirling disease is widely known in the United States as a serious fish health issue associated with economic and ecological impacts including salmonid population effects.

Impacts of the whirling disease parasite are highly variable. The introduction of *M. cerebralis* does not always result in severe wild trout population declines and there are many instances where the parasite has persisted for decades with no observable effect. When high numbers of parasites combine with susceptible fish, whirling disease can result in high mortality rates. In some locations, severe wild trout population declines have been attributed to whirling disease. Fish culture has also been significantly impacted by the parasite as its detection may require quarantine of a culture facility, destruction of fish stocks, disinfection, facility renovations, and even closure. Thus, significant economic impacts to fish culture have results from the necessary preventative actions and mitigation associated with whirling disease.

Whirling disease presents an informative case study in a variety of ways – as an exotic species, a fish pathogen, a perceived natural catastrophe, a perceived nonevent, a threat to wild and native salmonids, a risk to fish culture, and for extensive collaborative research and management. Since the discovery of *M. cerebralis,* research has expanded our understanding of this parasite, and also revealed how much there is still to learn. Research continues to contribute to a greater understanding of *M. cerebralis* disease ecology, host-parasite interactions, and invasive species ecology. This has made management strategies increasingly effective at protecting and enhancing salmonids in the wild and in fish culture. Current research is focused on understanding the dynamics of *M. cerebralis* ecology at a landscape level, characterizing the mechanisms of resistance in both hosts, refining risk assessments, and improving management tools. In 2009, a review of current research and management was completed which summarized the current status and knowledge on whirling disease and *M. cerebralis* (Elwell et al. 2009). The 2009 document focused on new research that revealed details of invertebrate host ecology, epidemiology among salmonids, management efforts and improved diagnostic techniques. Here we present some major topics from the 2009 document.

Disease Factors and Ecological Impacts

The clinical signs of whirling disease can be dramatic and include whirling behavior, blackened tail, skeletal deformities and mortality. These clinical signs are not unique to whirling disease, and can be caused by other conditions. Therefore, whirling disease cannot be diagnosed based on physical signs alone and must be confirmed by methods that specifically identify the parasite (see Elwell et al. 2009 for a review of diagnostic techniques). Infection and severity of disease is also highly variable. Factors influencing infection by *M. cerebralis* and the development of whirling disease include salmonid host factors, invertebrate factors, and environmental factors. Salmonid factors include the age and size of the salmonid host, the species and strain, and life history characteristics. Invertebrate host factors that may influence the infection and disease in salmonids include benthic community composition, the prevalence of infection in the *T. tubifex* host, and lineage and genetic variation of *T. tubifex*. Environmental factors strongly influence infection by *M. cerebralis* and the occurrence of whirling disease among salmonids. Environmental factors including water temperature, substrate, and flow can directly and indirectly influence the impacts of *M. cerebralis* on fish. These factors affect the parasite, its hosts, and the risk of disease, and may account for much of the variability observed in *M. cerebralis* detections and impacts.

Whirling disease can cause wild trout population declines, changes in fish community composition and, potentially, food chain effects. Wild trout population declines were first observed in North America in the upper Colorado River watershed, Colorado in 1993 (Walker and Nehring 1995). In some waters, declines were severe. In one 3.2-km reach of the Gunnison River, numbers of wild rainbow trout *Oncorhynchus mykiss* longer than 15 cm declined from approximately 11,000 to 86 fish between 1987 and 2003, a loss greater than 99% (Nehring 2006). Similar declines in wild rainbow trout were reported in at least five other Colorado rivers during this period (Nehring 2006). Population declines were also observed during this period in wild rainbow trout of the Madison River, Montana. In 1994, wild rainbow trout were reduced nearly 90% from historic averages in sections of the upper Madison River (Vincent 1996). These declines were especially marked among wild rainbow trout less than 2 years old.

More recently, declines among Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* in Yellowstone National Park, Wyoming, have been linked to whirling disease in combination with other factors, including drought and predation by illegally introduced lake trout *Salvelinus namaycush* (Koel et al. 2005a, b; Koel et al. 2006a, b; Murcia et al. 2006). Yellowstone cutthroat trout in Pelican Creek have been significantly affected by whirling disease. Pelican Creek once supported one of the largest spawning runs of Yellowstone cutthroat trout from Yellowstone Lake, with up to 30,000 upstream migrants per year (Koel et al. 2005a). High infection prevalence and disease severity among Pelican Creek cutthroat trout have contributed to a severe decline in this spawning population. Angling on Pelican Creek was completely closed in 2004, reports have described the spawning population as "essentially lost," and few wild-reared fry were observed between 2001 and 2004 (Koel et al. 2005a, b).

In Utah, declines in wild rainbow and cutthroat trout *Oncorhynchus clarkii* populations in several streams (e.g., the Beaver and Logan rivers) may be linked to *M. cerebralis* (Wilson 2006). Biologists also speculated there were wild rainbow trout population declines in the Big Lost River, Idaho due to a combination of whirling disease and drought; however, population impacts were not observed in other drainages where the parasite was present (Elle 1997). These and other wild trout population declines in the Intermountain West have led to concerns about how whirling disease could further affect native trout already threatened by habitat loss, invasive species, predation, and hybridization.

Whirling disease can lead to changes in fish community composition, as more resistant species such as brown trout *Salmo trutta* replace susceptible species. In several Montana drainages (Madison River, Rock Creek and Willow Creek), rainbow trout populations decreased while brown trout remained constant or increased (Baldwin et al. 1998; Granath et al. 2007). Similarly, brown trout numbers have increased in the Beaver River, Utah while wild rainbow have diminished in a pattern potentially linked to whirling disease (Wilson 2006). Reduced abundance of some wild salmonids could also affect food chains for fish-eating predators such as bears, eagles and other fishes (e.g., Koel et al. 2005b). Mountain whitefish *Prosopium williamsoni* are susceptible to *M. cerebralis* during their early life stages (Schisler et al. 2008), and if disease reduced this important forage fish, food chain impacts for predatory fish such as the threatened bull trout *Salvelinus confluentus* could be significant.

Population losses of the magnitude described above have not been consistently observed despite the widespread distribution of *M. cerebralis*. Population-level impacts of whirling disease have not been reported in the eastern and coastal-western United States despite the parasite being widely distributed in these regions. Modin (1998) noted the widespread occurrence of *M. cerebralis* in wild and cultured trout of California; however, wild trout population impacts were not detected. Conspicuous declines in wild trout population of Pennsylvania or the mid-Atlantic region as a result of whirling disease have not occurred, despite the presence of *M. cerebralis*, *T. tubifex*, and susceptible trout species (Schachte and Hulbert 1998; Hulbert 2005; Kaeser et al. 2006; Kaeser and Sharpe 2006).

Thus, the local and regional impacts of whirling disease in the United States remain variable and difficult to predict. Kaeser et al. (2006) speculated that outbreaks of whirling disease might have occurred among wild trout in Pennsylvania, but without sufficient information about fish abundance and infection dynamics from long-term monitoring programs, it's possible that whirling disease impacts could go unnoticed or unexplained. Additionally, the dynamics of the disease are complicated by the ecology of aquatic ecosystems, the parasite's two hosts, and the difficulty in accurately detecting the parasite. Wild trout declines in the Intermountain West have generated continuing concern about viable sport fisheries and the conservation of native salmonids.

Control and Mitigation

Once *M. cerebralis* is present in natural habitats where susceptible hosts coexist and environmental conditions are appropriate, it is generally considered impossible to eradicate. However, it is possible to control or slow further spread of the parasite and to mitigate its impacts. In fish culture facilities, there are many strategies to prevent the introduction of *M. cerebralis*, to disinfect the facility, and to control its spread. One example of successfully preventing spread of *M. cerebralis* from an infected hatchery to wild salmonid populations in Oregon is summarized by Bartholomew et al. (2007), who monitored infection in the adjacent creek before and following closure of earthen ponds on the facility. In that case, although the hatchery provided ample habitat for *T. tubifex*, the oligochaete was not abundant in the creek and partial closure of the hatchery removed the source of infection. In Maryland, a recent detection of *M. cerebralis* in hatchery fish prompted aggressive action by the Department of Natural Resources to prevent the spread of the parasite. This included testing all fish production facilities and destroying infected fish, the development of new biological security protocols, testing resident fish and surface waters regularly for the parasite and a new angler education program. These efforts suggest the parasite has remained localized in the area of original detection (Rivers 2008).

Fish culture strategies to deal with *M. cerebralis* include infrastructure alterations, application of chemicals for disinfection, and careful regulation and inspections. Advances in diagnostic techniques (see Elwell et al. 2009) for *M. cerebralis* should improve detection of the parasite in fish and water sources and reduce the likelihood of transferring infected fish, a major vector for the parasite. Wagner (2002) provides an excellent review of physical, chemical, and medical treatments tested for management of *M. cerebralis*. Physical modifications of facilities by converting earthen-bottom ponds and raceways to concrete to remove *T. tubifex* worm habitat have been very effective in reducing disease severity (Markiw 1992). A pathogen-free water supply must be secured to ensure eradication of the parasite. Unfiltered surface water sources may be vulnerable to *M. cerebralis* introductions by humans and wildlife. Many facilities are converting from surface water sources to groundwater sources to minimize exposure to water that may contain *M. cerebralis*. Incoming hatchery water sources can be treated by ozonation, chlorination, and ultraviolet light treatment to deactivate triactinomyxons (Markiw 1992; Hedrick et al. 1998; Hedrick et al. 2000). The use of filters to reduce incoming *M. cerebralis* has been less effective at minimizing infection among hatchery fish, but sand-charcoal filters and 10- to 20-μm Nitex cloth sieves can remove triactinomyxons (Hoffman 1962; Hoffman 1974; Wagner 2002; Arndt and Wagner 2003). Recent unpublished studies in Utah indicate that some commercial filtration systems that use a combination of three-dimensional filtration media and ultraviolet light are effective at removing triactinomyxons from hatchery water supplies (Arndt 2005). However, these options can be expensive to implement. If water sources cannot be effectively secured or treated, and the facility cannot be disinfected, outdated hatcheries may be decommissioned.

Several drugs have been tested in an effort to reduce or avoid infection in fish, but with little success (Hoffman et al. 1962; Taylor et al. 1973; Alderman 1986; El-Matbouli and Hoffmann 1991; Staton et al. 2002; Doyle et al. 2003; Clarkson et al. 2004). The utility of drugs in reducing infection prevalence in fish is limited, due to the complexity in Federal Drug Administration drug registration and approval, and difficulty for application with wild fish.

Regulations regarding fish health inspections and stocking of *M. cerebralis*-infected fish vary from state to state. For example, Montana and New York do not allow stocking of any infected fish into public waters; conversely, California and Colorado may allow stocking of infected fish into waters where the parasite has previously been detected. This practice is largely limited to waters not connected to any streams that can support wild salmonids, and the highest level of protection for wild salmonids is provided when there is no stocking or transfers of *M. cerebralis*-infected fish.

Control of *M. cerebralis* in the wild is more difficult than in culture facilities, but there are opportunities for decreasing disease impacts through fish stocking practices and stream modifications. In systems where fish are stocked, larger fish (> 40 mm, fork length) that are less susceptible to infection result in decreased myxospore numbers, although larger fish can still become infected (Ryce et al. 2004). Stocking larger fish is one of the most simple and most effective management strategies available. Stream restoration and stream habitat modification have been proposed as a means to reduce the impacts of whirling disease, but their benefits have been difficult to assess small-scale engineered stream modifications have been attempted, but the overall effectiveness in reducing incidents of infection and whirling disease in a stream system is unclear (Thompson and Nehring 2003, 2004; Waddle et al. 2006). Passive stream restoration may provide more effective enhancement of fish health. For example, livestock exclusion could lower stream temperatures by increasing riparian shading, and would reduce sediment and nutrients entering the stream. This could improve both fish health and reduce the risk of infection from *M. cerebralis* (Hansen et al. 2006). Finally, in regulated rivers, streamflow modifications such as flushing flows could serve to reduce infection severity by removing fine sediments and reducing *T. tubifex* habitat (Milhous 2005; Hallett and Bartholomew 2008).

Taking advantage of the variations in natural resistance between species of salmon and strains of trout can also be an effective management tool. Salmonids are highly variable in susceptibility to *M. cerebralis* and this resistance may be an innate characteristic of the species (e.g. coho salmon *Oncorhynchus kisutch*), or it may have evolved through natural selection in the presence of the parasite (e.g. brown trout, certain strains of rainbow trout). Preliminary research by E. R. Vincent (formerly of Montana Fish, Wildlife & Parks, personal communication) on the Madison River, Montana, suggests that progeny of wild rainbow trout that have survived the brunt of infection may have genetic resistance to infection. This resistance may be the result of natural variability, strong directional selection, or genetic mutation. Fostering wild fish populations with high genetic diversity may increase the likelihood of population persistence and resilience despite the presence of pathogens. The application of this knowledge as a management tool is uncertain but promising.

Additionally, a domestic strain of rainbow trout known as GR, which has reduced susceptibility to *M. cerebralis*, is being evaluated to determine the mechanisms of resistance and their potential to survive and reduce parasite levels in *M. cerebralis*-positive waters (Hedrick et al. 2003; Schisler et al. 2006; Wagner et al. 2006;). It is expected that *M. cerebralis*-resistant trout would have higher survival rates, successful reproduction, and could lower the number of parasites in the ecosystem by reducing the number of myxospores produced. By crossing this strain with locally adapted strains of rainbow trout, managers hope to develop a fish for stocking that would have resistance to whirling disease but retain genetic traits important for survival in the wild. The risks and benefits of this approach have been carefully considered and currently experiments are being conducted in three state hatchery programs (Colorado, Utah, and California). Initial results crossing resistant GR rainbow trout with locally adapted Colorado River rainbow trout show these fish have survived and reproduced in the wild following stocking into the Gunnison River, one of the Colorado rivers most heavily impacted by whirling disease.

A similar approach is being tested using an experimental introduction of resistant *T. tubifex* lineages into a Colorado stream to examine the effects of introduced resistant-lineage worms on the prevalence of infection and production of triactinomyxons (Winkelman et al. 2007; Thompson et al. 2008). Managers hope resistant *T. tubifex* could lower infection prevalence among susceptible *T. tubifex* through competitive interactions and by deactivation of myxospores by resistant *T. tubifex*. The extensive distribution and densities of aquatic worm populations and the complexity of natural ecosystems present challenges for this approach. As with resistant fish, the benefits and risks of introducing organisms into the wild will be carefully considered.

An important method for limiting the spread of *M. cerebralis* and minimizing its impacts is providing information to the public and engaging the public in resource management. Anglers, boaters and others must be aware of the potential role they play in the transport of *M. cerebralis*. Individuals with private fish ponds should be aware of fish health regulations, obtain any necessary permits before stocking fish into a pond, and ask to see a fish provider's fish health certification before purchasing fish. Several recommended precautions exist to prevent the spread of *M. cerebralis* and other aquatic invasive species: never transport live fish from one water body to another; rinse all mud and debris from equipment and wading gear; drain water before leaving the river or lake and allow boats and gear to dry between trips; do not use trout, whitefish, or salmon parts as bait; dispose of fish entrails and skeletal parts away from streams or rivers including disposing of salmonid fish parts in the garbage rather than a kitchen sink disposal.

Acknowledgements

We thank Dave Kumlien and the Trout Unlimited Whirling Disease Foundation along with the many biologists, managers and researchers who participated in the whirling disease research and provided assistance with the 2009 white paper.

Literature Cited

Alderman, D. J. 1986. Whirling disease chemotherapy. Bulletin of the European Association of Fish Pathologists 6:38-40.

Arndt, R. 2005. Pilot project at Midway Hatchery to evaluate several whirling disease filtration techniques. Utah Division of Wildlife Resources Fisheries Experiment Station Ichthyogram 16:1-3.

Arndt, R. E., and E. J. Wagner. 2003. Filtering *Myxobolus cerebralis* triactinomyxons from contaminated water using rapid sand filtration. Aquacultural Engineering 29:77-91.

Bartholomew, J. B., H. V. Lorz, S. D. Atkinson, S. L. Hallett. D. G. Stevens, R. A. Holt, K. Lujan, and A. Amandi. 2007. Evaluation of a management strategy to control the spread of *Myxobolus cerebralis* in a lower Columbia River tributary. North American Journal of Fisheries Management 27:542-550.

Clarkson, A. W., J. G. Rohrbough, and B. W. Hicks. 2004. Development of food-based vaccines for salmonids for inoculation against infectious hematopoetic necrosis virus and, potentially, whirling disease. Abstracts of Papers of the American Chemical Society 227:U427-U428.

El-Matbouli, M., and R. W. Hoffmann. 1991. Prevention of experimentally induced whirling disease in rainbow trout *Oncorhynchus mykiss* by Fumagillin. Diseases of Aquatic Organisms 10:109-113.

Elle, S. 1997. Distribution of *Myxobolus cerebralis* in Idaho with data presented for trout populations in four of the positive drainages. Pages 27-30 *in* Proceedings of the Whirling Disease Symposium, Logan, Utah, Whirling Disease Foundation, Bozeman, Montana.

Elwell, L. C. S., K. Stromberg, E. Ryce and J. Bartholomew. 2009. Whirling Disease in the United State: A Summary of Progress in Research and Management. Trout Unlimited Whirling Disease Foundation. Bozeman, Montana.

Granath, Jr., W. O., M. A. Gilbert, E. J. Wyatt-Pescador and E. R. Vincent. 2007. Epizootiology of *Myxobolus cerebralis*, the causative agent of salmonid whirling disease in the Rock Creek drainage of west-central Montana. Journal of Parasitology 93:104-119.

Hallett, S. L. and J. L. Bartholomew. 2008. Effects of water flow on the infection dynamics of *Myxobolus cerebralis*. Parasitology 135:371-384.

Hansen, E., P. Budy, and G. Theide. 2006. Evaluating the effectiveness of passive stream restoration for improving native fish health and reducing the impacts of whirling disease. Pages 44-46 *in* Proceedings of the 12th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Hedrick, R. P., M. El-Matbouli, M. A. Adkison, and E. MacConnell. 1998. Whirling disease: re-emergence among wild trout. Immunological Reviews 166:365-376.

Hedrick, R. P., T. S. McDowell, G. D. Marty, K. Mukkatira, D. B. Antonio, K. B. Andree, Z. Bukhari, and T. Clancy. 2000. Ultraviolet irradiation inactivates the waterborne infective stages of *Myxobolus cerebralis*: a treatment for hatchery water supplies. Diseases of Aquatic Organisms 42:53-59.

Hedrick, R. P., T. S. McDowell, G. D. Marty, G. T. Fosgate, K. Mukkatira, K. Myklebust, and M. El-Matbouli. 2003. Susceptibility of two strains of rainbow trout (one with suspected resistance to whirling disease) to *Myxobolus cerebralis* infection. Diseases of Aquatic Organisms 55:37-44.

Hoffman, G. L. 1962. Whirling disease of trout. United States Department of the Interior, Fish and Wildlife Service, Fishery Leaflet 508, Washington, D.C.

Hoffman, G. L. 1974. Disinfection of contaminated water by ultraviolet irradiation, with emphasis on whirling disease (*Myxosoma cerebralis*) and its effect on fish. Transactions of the American Fisheries Society 103:541-550.

Hoffman, G. L., C.E. Dunbar, and A. Bradford. 1962. Whirling disease of trouts caused by *Myxosoma cerebralis* in the United States. United States Fish and Wildlife Service, Special Scientific Report, Fisheries No. 427.

Hulbert, P.J. 2005. Whirling Disease. New York State Conservationist. New York State Department of Environmental Conservation, Albany.

Kaeser, A. J., and W. E. Sharpe. 2006. Patterns of distribution and abundance of *Tubifex tubifex* and other aquatic oligochaetes in *Myxobolus cerebralis* enzootic areas in Pennsylvania. Journal of Aquatic Animal Health 18:64-78.

Kaeser, A. J., C. Rasmussen and W. E. Sharpe. 2006. An examination of environmental factors associated with *Myxobolus cerebralis* infection of wild trout in Pennsylvania. Journal of Aquatic Animal Health 18:90-100.

Koel, T. M., J. L. Arnold, P. E. Bigelow, P. D. Doepke, B. D. Ertel, and D. L. Mahony. 2005a. Yellowstone Fisheries and Aquatic Sciences Annual Report, 2004. YCR-2005-04, National Park Service, Yellowstone Center for Resources, Yellowstone National Park, Wyoming.

Koel, T. M., P. E. Bigelow, P. D. Doepke, B. D. Ertel, and D. L. Mahony. 2005b. Nonnative lake trout result in Yellowstone cutthroat trout decline and impacts to bears and anglers. Fisheries 30:10-19.

Koel, T. M., J. L. Arnold, P. E. Bigelow, P. D. Doepke, B. D. Ertel, D. L. Mahony, and M.E. Ruhl. 2006a. Yellowstone Fisheries and Aquatic Sciences Annual Report, 2005. YCR-2006-09, National Park

Service, Yellowstone Center for Resources, Yellowstone National Park, Wyoming.

Markiw, M. E. 1992. Salmonid whirling disease. Leaflet 17, United States Fish and Wildlife Service, Washington, D.C.

Milhous, R. 2005. Management of fine sediments in river channels. Pages 32-33 *in* Proceedings of the 11th Annual Whirling Disease Symposium, Denver, Colorado. Whirling Disease Foundation, Bozeman, Montana.

Modin, J. 1998. Whirling disease in California: A review of its history, distribution, and impacts, 1965-1997. Journal of Aquatic Animal Health 10:132-142.

Murcia, S., B. L. Kerans, E. MacConnell, and T. Koel. 2006. *Myxobolus cerebralis* infection patterns in Yellowstone cutthroat trout after natural exposure. Diseases of Aquatic Organisms 71:191-199.

Nehring, R. B. 2006. Colorado's cold water fisheries: Whirling disease case histories and insights for risk management. Special Report No. 79, Colorado Division of Wildlife, Aquatic Wildlife Research. Fort Collins, Colorado.

Rivers, S. E. 2008. The status of whirling disease in Maryland, 2008. Pages 15-16 *in* Proceedings of the 14th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Ryce, E. K. N., A. V. Zale, and E. MacConnell. 2004. Effects of fish age and development of whirling parasite dose on the disease in rainbow trout. Diseases of Aquatic Organisms 59:225-233.

Schachte, J. H. and P. J. Hulbert. 1998. A summary of field research and monitoring for *Myxobolus cerebralis* in New York state from 1994 to 1997. Pages 17-19 *in* Proceedings of the 4th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Schisler, G. L., K. A. Myklebust and R. P. Hedrick. 2006. Inheritance of *Myxobolus cerebralis* resistance among F_1-generation crosses of whirling disease resistant and susceptible rainbow trout strains. Journal of Aquatic Animal Health 18:109-115.

Schisler, G. L., K. Rogers, and R. P. Hedrick. 2008. Early development of mountain whitefish (*Prosopium williamsoni*) and effects of *Myxobolus cerebralis* exposure. Pages 21-22 *in* Proceedings of the 14th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Staton, L., D. Erdahl, and M. El-Matbouli. 2002. Efficacy of Fumagillin and TNP-470 to prevent experimentally induced whirling disease in rainbow trout *Oncorhynchus mykiss*. Pages 239-247 *in* J. L. Bartholomew and J. C. Wilson, editors. American Fisheries Society Symposium 29, Bethesda, Maryland.

Taylor, R. E. L., S. J. Coli, and D. R. Junell. 1973. Attempts to control whirling disease by continuous drug feeding. Journal of Wildlife Diseases 9:302-305.

Thompson, K. G. and R. B. Nehring. 2003. Habitat modifications to reduce *Myxobolus cerebralis* infection in streams: A preliminary report. Page 56 *in* Proceedings of the 9th Annual Whirling Disease Symposium, Seattle, Washington, Whirling Disease Foundation, Bozeman, Montana.

Thompson, K. G. and R. B. Nehring. 2004. Evaluating the efficacy of physical habitat modification to reduce the impacts of *Myxobolus cerebralis* infection in streams. Pages 46-47 *in* Proceedings of the 10th Annual Whirling Disease Symposium, Salt Lake City, Utah, Whirling Disease Foundation, Bozeman, Montana.

Thompson, K. G., D. Winkelman, and C. Clapp. 2008. Investigating the potential for biological control of whirling disease in natural streams through introductions of resistant *Tubifex tubifex*. Pages 1-2 *in* Proceedings of the 14th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Vincent, E. R. 1996. Whirling disease and wild trout: The Montana experience. Fisheries 21:32-33.

Waddle, T. J., and J. W. Terrell. 2006. First year performance of permeable tubifex habitat isolation berms on the Cache la Poudre River. Pages 42-43 *in* Proceedings of the 12th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Wagner, E. 2002. Review: Whirling disease prevention, control, and management: A review. Pages 217-225 *in* J. L. Bartholomew and J. C. Wilson, editors. Whirling disease: Reviews and current topics. American Fisheries Society Symposium 29, Bethesda, Maryland.

Wagner, E. J., C. Wilson, R. Arndt, P. Goddard, M. P. Miller, A. Hodgson, and R. Vincent. 2006. Evaluation of disease resistance of the Fish Lake-DeSmet and Harrison Lake strains of rainbow trout exposed to *Myxobolus cerebralis*. Journal of Aquatic Animal Health 18:128-135.

Walker, P. G. and R. B. Nehring. 1995. An investigation to determine the cause(s) of the disappearance of young wild rainbow trout in the Upper Colorado River, in Middle Park, Colorado. Colorado Division of Wildlife. Fort Collins, Colorado.

Wilson, J.C. 2006. Whirling disease: What we've learned in the last 15 years. Wildlife Review. Utah Division of Wildlife Resources, Summer 2006. Logan, Utah.

Winkelman, D., C. Clapp, and K. Thompson. 2007. Investigating competition among lineages of *T. tubifex* and the potential for biological control of whirling disease in natural streams. Pages 28-29 *in* Proceedings of the 13th Annual Whirling Disease Symposium, Denver, Colorado, Whirling Disease Foundation, Bozeman, Montana.

Session 5: Management and Conservation of Wild Trout

SPAWNING MIGRATION PATTERNS OF NATIVE CUTTHROAT TROUT IN THE UPPER SNAKE RIVER

Kristen Homel[1] and Robert E. Gresswell[2]

[1] Ph.D. Graduate Research Assistant, Montana Cooperative Fishery Research Unit, Montana State University, P.O. Box 173460, Bozeman, MT 59715, Phone: 406.994.1823, Fax: 406.994.7479, Email: kristen.homel@gmail.com

[2] Research Biologist, U.S. Geological Survey, Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, MT 59715, Phone: 406.994.7085, Fax: 406.994.6556, Email: bgresswell@usgs.gov

ABSTRACT — Potamodromous cutthroat trout *Oncorhynchus clarkii* express multiple life-history strategies that appear to be directly related to the physical habitat in which they evolved (Gresswell et al. 1994). Although there is a substantial amount of information concerning cutthroat trout that live in lakes and ascend tributaries to spawn (lacustrine-adfluvial), information about populations in lotic environments is less abundant, and most studies have focused on the migration and spawning dynamics of cutthroat trout moving into tributaries (fluvial-adfluvial; Brown and Mackay 1995). In larger river systems, however, little is known about the spatiotemporal relationships between physical characteristics of the habitat and life-history organization of cutthroat trout. Given that harvest and predation in river networks are spatially structured (Schlosser 1991), it is important to understand the non-spawning distribution of individuals from different spawning populations or with different reproductive strategies, to protect biodiversity and maintain persistence of native fishes.

In the upper Snake River, Snake River finespotted cutthroat trout *Oncorhynchus clarkii behnkei* provide a unique opportunity to evaluate the life-history organization of a native cutthroat trout in a large connected river network. Snake River finespotted cutthroat trout are the primary form of cutthroat trout found in the main stem of the Snake River upstream of Palisades Reservoir (which comprises the majority of the native range of the subspecies) and express multiple life-history strategies. Although two large dams and many small impoundments exist in the upper Snake River watershed, a substantial amount of connected habitat remains in the stream network. To this end, radio telemetry was used to determine (1) the locations and spatial structure of Snake River finespotted cutthroat trout spawning populations, (2) effects of physical characteristics of the stream segment where cutthroat trout were tagged on the migration distance from tagging site to spawning site, and (3) physical characteristics of the Snake River that influence the relative abundance of spawners per segment.

Initially, 248 cutthroat trout were implanted with radio tags (Figure 1) in September and October of 2007 (n = 49) and 2008 (n = 199), and individual movements were monitored for a year. Preliminary relocation of cutthroat trout occurred two weeks after radio-tagging. Relocations occurred bimonthly in November and January, weekly from April to July, and biweekly from August to October. Concurrently, physical habitat surveys were conducted in segments of the Snake River between Jackson Lake Dam and Moose Junction. Five segments were delineated according to major tributary junctions (*sensu* Frissell et al. 1986): Jackson Lake Dam to Pacific Creek (segment A), Pacific Creek to Spread Creek (segment B), Spread Creek to Deadman's Bar (segment C), Deadman's Bar to Cottonwood Creek (segment D), and Cottonwood Creek to Moose Junction (segment E; Figure 1).

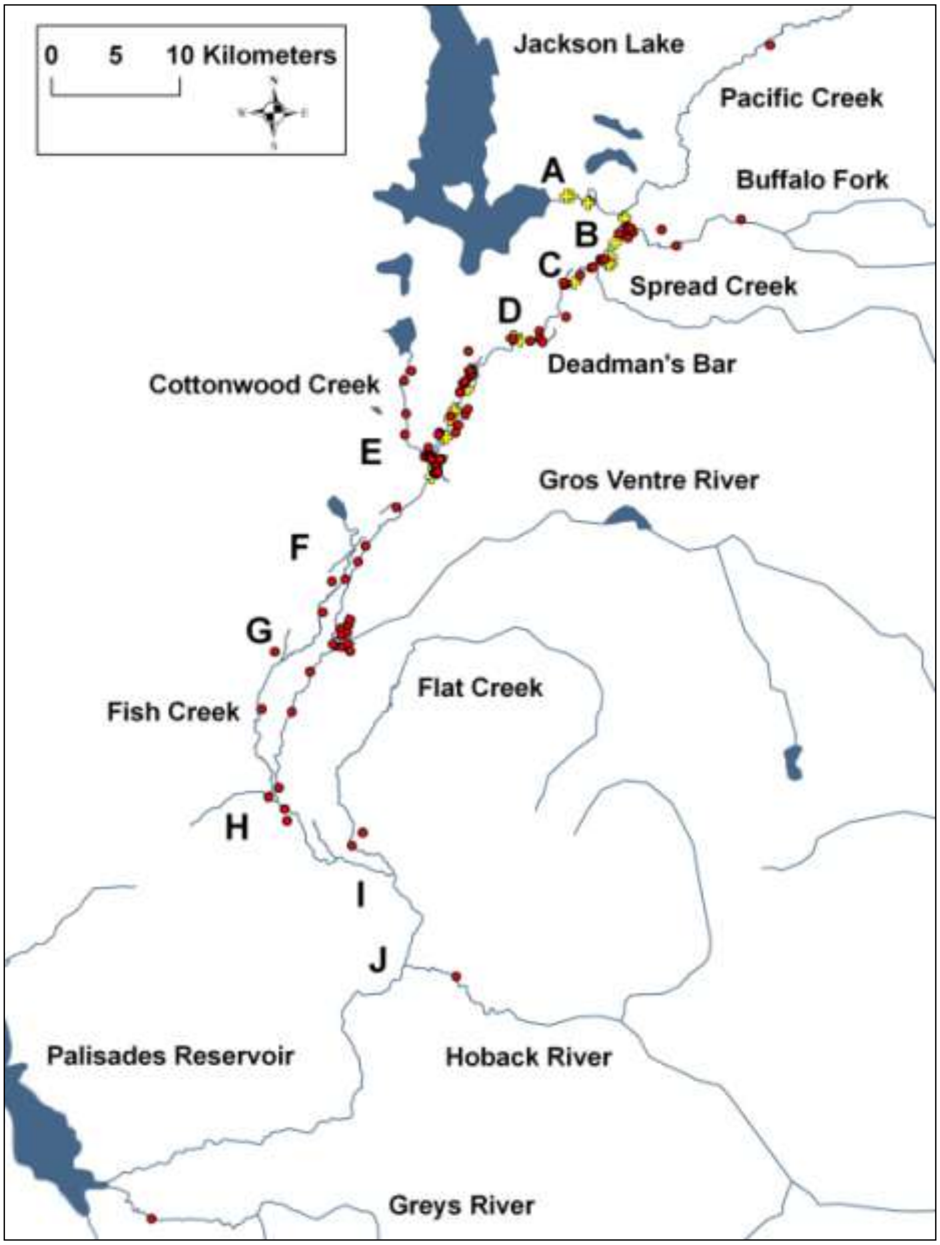


Figure 1. Snake River study area from Jackson Lake Dam to the Palisades Reservoir, 2009. Major tributaries are labeled and the top of each sample segment is marked with the segment label. Segment A begins at Jackson Lake Dam; segment B begins at Pacific Creek, segment C begins at Spread Creek, segment D begins 800 m upstream of Deadman's Bar, segment E begins at Cottonwood Creek, segment F begins at the top of the levy, segment G begins at the Gros Ventre River, segment H begins at Fish Creek, segment I begins at Flat Creek, and segment J begins at the Hoback River and extends to Palisades Reservoir. Radio tagging locations of 248 Snake River finespotted cutthroat trout are indicated by crosses. Spawning locations of 145 radio-tagged Snake River finespotted cutthroat trout indicated by dots.

Snake River finespotted cutthroat trout spawning occurred in 33 tributaries, mainstem side channels, and spring-creek systems in the upper Snake River watershed (Figure 1). A total of 145 spawning migrations was recorded in 2008 and 2009, including seven cutthroat trout that spawned in both years. Of these migrations, 31 cutthroat trout migrated upstream in the Snake River to a spawning area or confluence with a spawning tributary, and 114 cutthroat trout migrated downstream to spawn. The majority of spawning activity occurred within 45 km of the location where individuals were originally tagged, and the longest migration was >100 km to the confluence of the Grey's River. Both fluvial-adfluvial and fluvial (trout that spawn and rear in the same river) migration patterns were observed. Fluvial-adfluvial cutthroat trout spawned in tributaries (n = 40) and spring creeks (n = 62), and fluvial cutthroat trout spawned in side channels or the main stem of the Snake River (n = 43). Only eight Snake River finespotted cutthroat trout migrated to the spawning area nearest to the location of tagging, but 26 individuals entered the nearest spawning area of the type that corresponded to the spawning strategy (e.g., spring creek fluvial-adfluvial spawner).

Spawning migration distance varied in relation to tagging location and the life-history strategy of the individual. For example, Snake River finespotted cutthroat trout that were tagged in segments A, B, and C moved significantly farther to access spawning areas (average migration distance = 28.1 km) than those radio-tagged in segments D and E (average migration distance = 14.9 km; P <0.01, Figure 2). Spawning migration distance also differed significantly among life-history strategies. Fluvial-adfluvial cutthroat trout migrated substantially farther along the Snake River to access the spawning area confluence (average migration distance = 22.9 km for spring creek spawners and 30.5 km for tributary spawners) than fluvial cutthroat trout (average migration distance = 8.5 km; $P < 0.01$).

Not only did spawning migration distance vary among segments, but the total number of spawners within a segment varied significantly. A total of 100 out of 145 cutthroat trout spawned (or accessed tributary and spring creek spawning areas) in segments A-E in the Snake River. There was a statistically significant difference in the abundance of spawners per segment ($P < 0.01$). Additionally, the abundance of spawners per segment was positively and significantly associated with the abundance of large woody debris in a segment (r^2 = 0.82, P = 0.02).

Snake River finespotted cutthroat trout spawned in many portions of the upper Snake River network. The spawning migration distances (both upstream and downstream) observed in this study, coupled with the observation that cutthroat trout did not use

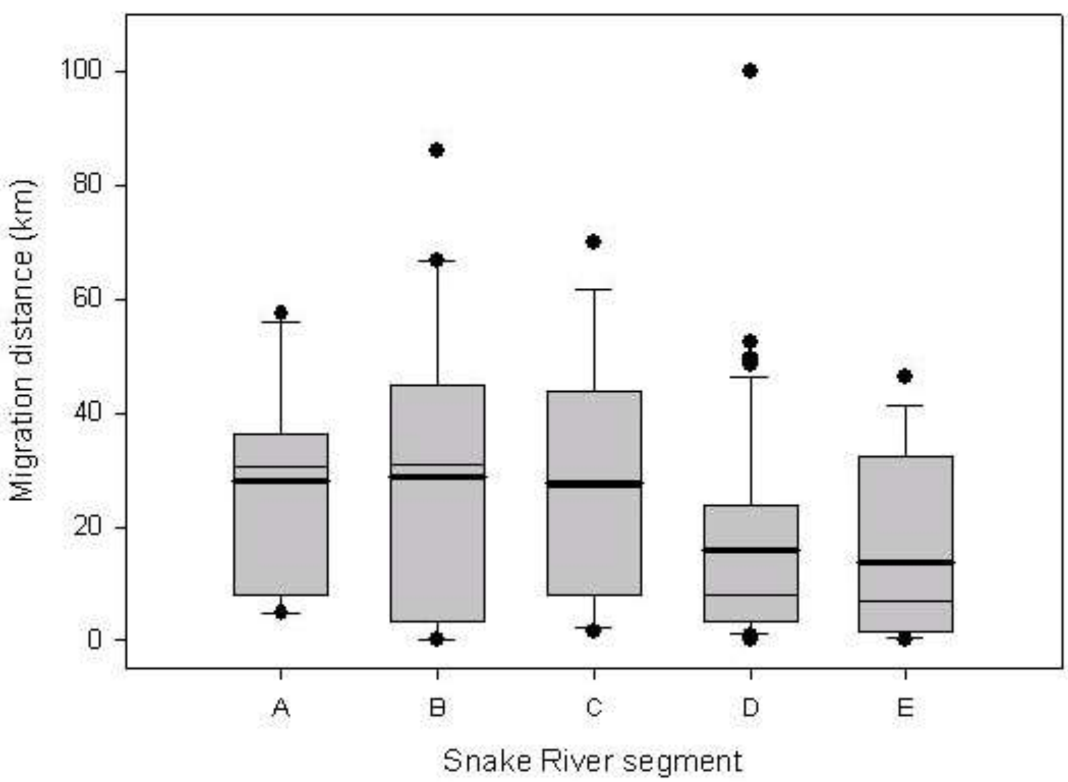


Figure 2. Spawning migration distance (km) from tag location to spawning location for fish originally tagged in segments A-E in the Snake River. Boxes represent observations between the 25th and 75th percentile and whiskers represent observations within the 10th and 90th percentile. The thin black line within the box is the 50th percentile and the thick black line is the mean migration distance.

the closest spawning area suggest that spawning location cannot be predicted from tagging location. However, spatial structure in distribution patterns was observed; individuals that spawned in the same locations tended to co-occur in the Snake River both before and after spawning. Snake River finespotted cutthroat trout exhibit complex behavioral strategies that connect spawning habitat in the watershed with movements across a range of spatiotemporal scales. The Snake River is not pristine, but it provides an excellent example of the importance of intact stream networks for supporting a full complement of behavioral patterns.

Acknowledgements

Many partners collaborated and volunteered on this project, and without their help we could not have accomplished our research objectives. Pat Connolly, Brady Allen, and Philip Haner (U.S. Geological Survey, Columbia River Research Laboratory) provided field equipment, radio telemetry training, and technical support. Rob Gipson, Tracy Stephens, and Diana Sweet (Wyoming Game and Fish) assisted with fish capture and tagging, supplied field equipment and technicians for field sampling, and gave input into the research design. Sue O'Ney and Cindy O'Neil (Grand Teton National Park) furnished research permits and access to Grand Teton National Park resources. Joe Neal (U.S. Forest Service) lent sampling equipment to us. Finally, many volunteers and technicians provided substantial time and expertise to our sampling efforts. Funding and in-kind support were provided by the National Fish and Wildlife Foundation One Fly program (grant numbers 2006-0095-007 and 2007-0074-010), the U.S. Geological Survey (grant number 8348-99I34), Wyoming Game and Fish, Grand Teton National Park, Montana Cooperative Fishery Research Unit, and the University of Wyoming-National Park Service AMK research station.

References

Brown, R. S., and W. C. Mackay. 1995. Fall and winter movements of and habitat use by cutthroat trout in the Ram River, Alberta. Transactions of the American Fisheries Society 124:873-885.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A hierarchical classification for stream habitat classification: viewing streams in a watershed context. Environmental Management 10: 199-214

Gresswell, R. E., W. J. Liss, and D. L. Larson. 1994. Life-history organization of Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*) in Yellowstone Lake. Canadian Journal of Fisheries and Aquatic Sciences 51 (Suppl. 1):298-309.

Schlosser, I. J. 1991. Stream fish ecology: a landscape perspective. BioScience 41:704-712.

Abundance and Effective Population Size (N_E) of Redband Trout Populations in Desert Streams of Southwest Idaho

Daniel J. Schill & George W. LaBar
Department of Fish and Wildlife Resources, University of Idaho, Moscow, ID

ABSTRACT — Drainage-wide abundance metrics and distribution patterns for redband trout *Oncorhynchus mykiss gairdneri*, a subspecies of special concern residing in arid southern Idaho basins, have not been investigated. In this study we used a combination of field density estimates (n = 595), models predicting adult maturity, and computer simulations to quantify distribution, estimate total population size, adult abundance (N_{Adult}) and effective population size N_e for redband trout populations in southwest Idaho river basins. There were nearly one million (925,000 ± 595,000) redband trout residing in the six Geographic Management Units during the study period, an estimate that should be viewed as conservative, given the known negative sampling bias of the enumeration techniques used. We concluded there were 34 redband trout subpopulations in the study area. Total subpopulation abundance estimates (fish ≥ 100 mm) were made for 15 subpopulations that ranged in size from 94 to 101,333 fish. The ratio of effective population size (N_e) to mean adult number (N_{Adult}) in simulated populations ranged from 0.37 to 0.82 over a 200- year simulation period. We estimated N_e in 14 of 34 subpopulations and obtained values ranging from 38 to about 41,000 with all but three estimates exceeding 1,000. While we are unable to make final conclusions regarding redband trout population abundance estimates, number of subpopulations, and N_e estimates across the southwest Idaho desert until an improved assessment of potential genetic introgression is completed, study results suggest that existing populations are not at appreciable risk in the near term. We recommend that an annual trend monitoring program be developed for representative redband trout subpopulations across the study area.

Introduction

Existing population density and distribution information for redband trout *Oncorhynchus mykiss gairdneri* residing in xeric basins across southern Idaho are sparse. There have been occasional density estimates obtained via electrofishing inventories (e.g. Allen et al. 1995), but such past sampling sites number less than 75, span a time period of over 25 years, and are spread across a vast expanse. Drainage-wide distribution patterns for redband trout across southern Idaho have not been systematically investigated in the field. Despite the lack of such abundance data and a scarcity of life-history information, redband trout in these streams were petitioned for protection under the Endangered Species Act in 1995, a petition found to be unwarranted at the time. Because of the relative risks faced by small populations *versus* large ones, substantial effort in conservation biology is devoted to quantification of current population size (McElhaney et al. 2000). Before any objective evaluation of the status of desert redband trout in Idaho can be made, a study quantifying densities, distribution, and total population size in various drainages is needed.

While the most important variables for evaluation of overall species status remain population abundance and trend (McElhaney et al. 2000), evaluation of genetic risk has also become common in the conservation arena (e.g. Mace and Lande 1991; Allendorf et al. 1997). Genetic guidelines for small population status typically rely on estimation of the effective population size or N_e. Effective population size is the size of an "ideal" population that would experience the same rate of genetic change as the actual population under consideration (Wright 1969).

Although there is general consensus on the importance of estimating N_e for making long-term management decisions, it is a challenging population parameter to obtain. Both genetic methods and direct demographic approaches have been used. However, demographic estimation often involves parameters that are difficult to obtain such as lifetime family size (Harris and Allendorf 1988) while genetic approaches can suffer from resolution issues for all but the smallest populations (Rieman and Allendorf 2001). Regardless of such difficulties, the 50/500 N_e rule has emerged as a widely applied "rule-of-thumb" for evaluating genetic risk for populations over the past several decades.

Because population abundance estimates, expressed as either total population size (N_{Census}) or adult abundance (N_{Adult}) are often the only data available for many populations, the ratio of N_e to either abundance metric (N_e/N) is conceptually an important variable for monitoring genetic diversity within populations (Frankham 1995). Taxon-specific N_e/N ratios would make it possible to predict ongoing genetic loss in populations by simply estimating N . Because of the costs and difficulties inherent in obtaining accurate estimates of such ratios, simulation modeling has been used as an alternative to field estimation (Harris and Allendorf 1988; Rieman and Allendorf 2001). The latter authors related estimates of N_e to N_{Adult} for a range of hypothetical bull trout *Salvelinus confluentus* populations, resulting in N_e/N_{Adult} approximations for fishery managers to apply directly to adult bull trout abundance estimates.

In this study, we estimate several population abundance parameters and distribution for redband trout. Specific study objectives were (1) estimate total population size, adult breeder abundance (N_{Adult}) and distribution of redband trout in various major southwestern Idaho river basins and connected tributaries in surrounding states, (2) derive a range of likely N_e/N_{Adult} ratios via simulation, and (3) estimate N_e in as many subpopulations as practical.

Study Area

The study area encompasses the Snake River basin upstream of the Boise River confluence to the natural redband trout barrier of Shoshone Falls, excluding the Big Wood River drainage (Figure 1). To facilitate estimation of redband trout population sizes, we divided the study area into six Geographic

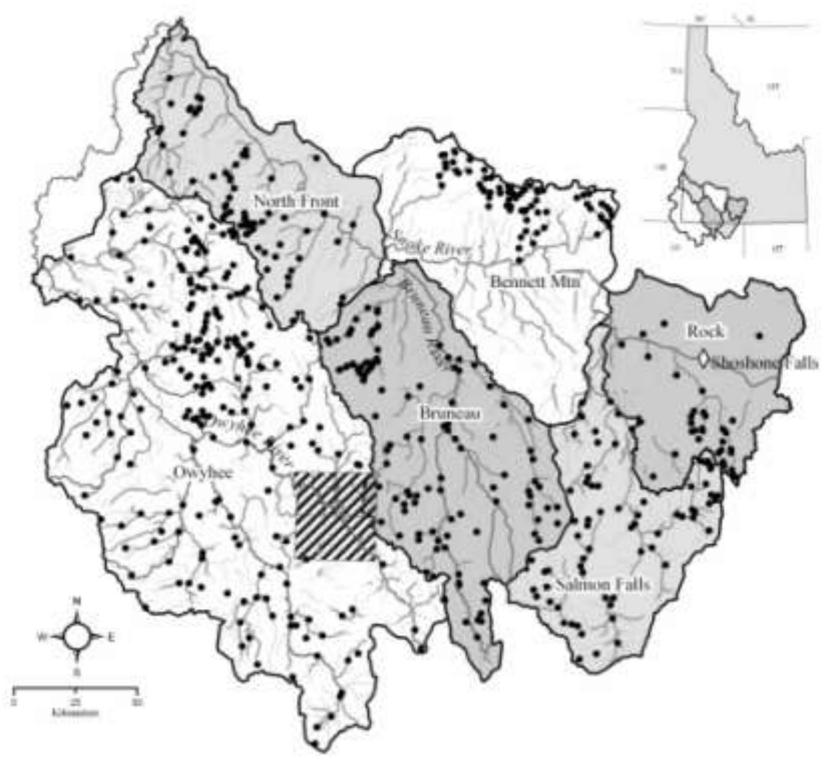


Figure 1. Distribution of six geographic management units (GMUs) and 595 randomly selected study sites (dots) of redband trout across desert river basins in southwestern Idaho. The shaded area with diagonal hatch marks indicates the area managed by the Sho-Pai Indian Reservation, and the diamond symbol is upper range boundary of Shoshone Falls.

Management Units or GMUs. Management unit boundaries were selected based largely on major river drainages, likely historical distribution, or known movement patterns. For a description of rainfall, streamflow, vegetation type, and fish species present, see Schill (2009).

METHODS

Redband Trout Abundance

Study Site Selection

To develop a sampling frame, biologists from the Bureau of Land Management and Idaho Fish and Game met and examined all stream courses on 1:100,000 maps of the study area and provided *a priori* distribution ranks for redband trout based on past experience or professional judgment. All stream reaches were coded for redband trout presence as "likely present", "likely absent", or "unknown".

To estimate abundance at the GMU scale, study sites were drawn from a GIS-layer within the above three strata with the assistance of the Environmental Protection Agency's Environmental Monitoring and Assessment Program design or EMAP (Stevens and Olsen 2004). Sample draws were stratified in two ways, the first stratification consisting of the three *a priori* distribution ranks noted above. To minimize variance of subsequent population abundance extrapolations, stream segments within strata where redband trout were likely present were 1.8 times more likely to be sampled as those segments where presence or absence was unknown and 22 times more likely to be sampled than those stream segments where redband trout were likely absent. Within these strata, stream order was used as a secondary stratification.

This sample site selection process was also used for abundance estimation at the subpopulation level (see below for definition) wherever possible. However, in some small, and isolated drainages in the Bennett Mountain and North Front GMU, a study site selection process with increased sample frequency per stream kilometer was used (see Meyer et al. 2006) to ensure adequate sample size for subpopulation abundance extrapolations.

Fish Sampling

Fish sampling occurred between 1999 and 2005 at base flow to moderate flow conditions, typically after spring runoff, from June to October. At nearly all study sites, fish were captured with DC electrofishing gear. Electrofishing sites were typically 80-120 m in length (mean = 92 m, range = 25-170 m). See Schill (2009) for a detailed description of electrofishing approach, fish handling procedures, and trout density estimation.

At the few sites where the stream channel was too large for removal electrofishing (n = 18), multiple snorkelers (Schill and Griffith 1984) were used to observe and count redband trout. Two snorkelers began at the bottom of sites and moved upstream in tandem recording redband trout abundance in 100-mm size classes. Direct counts were summed and considered to be total abundance estimates.

The area sampled by either electrofishing or snorkeling was estimated at all study sites. Stream length (m) was measured along the thalweg and mean stream width (nearest 0.1 m) was estimated from ten equally spaced transects within each site. Both electrofishing and snorkeling abundance estimates were converted to linear (fish/100 m) and areal (fish/100 m^2) density values using the above habitat measurements. Stream order was determined from 1:100,000 scale BLM land status maps.

Abundance Extrapolation

The above methodology resulted in a total of 595 randomly selected density estimates across the study area (Figure 1). Eighty percent of all stream kilometers in the study area were ranked as redband trout "likely absent". Only three of 139 sites (2 %) that fell in this stratum subsequently had redband trout present and their densities were quite low (mean = 0.3 fish/km), values considered numerically insignificant. Therefore, densities of trout in this stratum were not extrapolated across the landscape or considered further. For each GMU and remaining distribution stratum ("likely present" and "unknown"), we estimated total redband trout abundance and associated 90% confidence intervals separately by stream order using the formulas of Scheaffer et al.(1996) as described by Schill (2009). All sample sites, including fishless and dry sites, were included in the estimates (Meyer et al. 2006)

Within GMUs, we identified individual redband trout subpopulations based on our sampling results, local biologist knowledge regarding annual stream channel desiccation and connectivity, as well as information from a companion genetics study of redband trout population structure in the study area (IDFG Genetics Lab, unpublished data). Subpopulations were defined as populations that were physically disconnected from, or not experiencing substantial gene flow with other populations within a GMU (Meyer et al. 2006). For a more detailed description of subpopulation boundary definition see Schill (2009).

N_e Background and Estimation

Effective population size, N_e , is a theoretical construct with constant population size, equal lifetime family size, equal sex ratio, random mating, and discrete generations (Frankham 1995). Rieman and Allendorf (2001) tracked the loss of genetic variation in relation to five ranges of total adults "present" in a series of computer-simulated bull trout populations. Use of such a simulation approach to estimate N_e for plausible life history inputs allows concurrent, precise tracking of adult population abundance parameters difficult to measure in the wild (Harris and Allendorf 1989). This approach facilitated estimation of N_e / N_{Adult} ratios useful for approximation of bull trout N_e across the landscape (Rieman and Allendorf 2001). Below we describe the nearly identical approach used for estimation of the same ratio for desert redband trout populations.

Model Description

The software program VORTEX is an age-structured model that incorporates demographic stochasticity and environmental variation into simulated populations while tracking genetic change by following the history of individuals (Lacy et al. 2005).

Life History Inputs

Life history data, including median age of first breeding by sex, spawning ages, and age 1+ survival rates reported by Schill (2009) served as the primary data source for modeling efforts (Table 1). These data span the types of life history combinations observed across the study area (short *versus* long-lived, low to high natural mortality, identical maturation schedules by sex versus early maturation in males or females, etc).

The inability of the VORTEX model to readily accommodate high fecundities associated with fish populations for input as brood sizes is a stated limitation in the users manual (Miller and Lacy 2005). To input mean brood sizes into the model, we collapsed information from two life stages (egg to fry) assuming a 10% egg to fry survival (Thurow 1987; Palm 2007).

Given mean natural mortality rates of age-1 fish and maturity schedules estimated from a range of actual populations (Table 1), mean survival of fry to age-1 (age-0 mortality) was varied in the model to produce a stable population (r = 0) in all simulations.

Modeling Approach and outputs

We used the same overall modeling approach as Rieman and Allendorf (2001). Demographic stochasticity around reproduction and mortality are automatically modeled in VORTEX as binomial distributions (Miller and Lacy 2005). In addition to the range of life history values reported for redband trout (Table 1), three additional factors considered important in reducing N_e / N ratios, were all addressed in the model scenarios, including variable population size (VPS), variable reproductive success (VRS), and breeder sex ratio (SR) (Frankham 1995; Ardren and Kupuscinski 2003). Sex ratio was varied via the differing age-at-maturity schedules considered above.

Table 1. Range of life history parameters used to approximate the range of N_e/N ratios for desert redband trout in southwest Idaho.

Simulation	Stream name[a]	Longevity	Spawning ages[b] M	Spawning ages[b] F	Age 1+survival	Mean generation time
S1	Bennett/McMullen Creeks	5	2-5	3-5	0.43	3.05
S2	Castle Creek	6	3-6	4-6	0.49	4.13
S3	Duncan/Little Canyon Creeks	5	3-5	3-5	0.43	3.49
S4	Sinker Creek[c]	3	3	2-3	0.24	2.19
S5	Crab Creek	6	4-6	5-6	0.49	4.95
S6	Little Jacks Creek	9	6-9	5-9	0.76	6.87
S7	Castle Creek w/differential sex mortality	6	3-6	4-6	0.55 Males 0.49 Females	4.19

[a] Estimates for life history parameters taken from one or more streams as reported in Schill (2009) meant to encompass the range in the life history values for desert redband populations in Idaho.

[b] First ages are median age of first spawning.

[c] Assumed age-3 fish present in the population based on scales and length data.

Different methods were used to impart VRS for males and females. For females we used the ratio of the standard deviation (SD) of mean family size divided by mean family size from the only two useable datasets available in the salmonid literature. This ratio was 0.7 in Garant et al. (2001) for Atlantic salmon fry 3 months after hatching and 0.9 for resident brook trout at age 1-2 (Thériault et al. 2007a). The mean ratio from these two studies (0.8) was used to scale SD around mean female fry brood size in the simulated populations.

For males, VORTEX provides a mate monopolization feature that automatically imparts social structure in polygynous mating systems and excludes some adult males from the pool of available breeders each year. The model assumes male family size is described by a Poisson distribution, a reasonable assumption for salmonids given the recent results of Thériault et al. (2007b) for resident brook trout. The VORTEX-provided estimate of spawning success averaged 63% among males in the simulations.

Trend data monitoring in desert redband trout populations is limited, and there are no estimates of variance in natural mortality or survival due to environmental variation. A relatively small amount of variance in age-1+ mortality was included in model scenarios (SD = 2.5), while considerably more variation in age-0 mortality was incorporated (see Schill 2009 for more details).

To evaluate VPS, initial population size in each scenario was varied to produce five different average adult population sizes across the simulation period ranging from about 50 to 550 fish (50-75,100-150,200-250,300-375,450-550) and all simulations were run for 200 years with each initial starting population size replicated 500 times (Rieman and Allendorf 2001).

Upon completion of each simulation, we recorded four VORTEX output variables and the ratio of N_e/N_{Adult} was calculated for all scenario population sizes and subsequently plotted (see Schill 2009 for more detail). The range in these results was used below to approximate N_e for redband trout across the landscape.

Estimation of Mature Trout (N_{Adult}) and N_e Approximation in Subpopulations

The number of breeding-sized redband trout residing in GMUs and subpopulations was estimated using length frequency data obtained during field abundance estimation at the individual study sites (see fish sampling section above) and the approach of Meyer et al. (2006). We used logistic regression models relating the variable stream order to male and female length-at-maturity (see Schill 2009; Table 2.5 for formulas) to predict, at any given study site, the length at which the probability of a redband trout being mature was 0.5, hereafter termed the maturity transition point (MTP). At each fish sampling site, the length frequency of fish collected was compared with estimates of MTP at the site for both males and females to estimate how many of the trout were mature (Meyer et al. 2006). For fish with lengths between the MTP for males and females, we assumed the overall sex ratio was 50:50 (Schill 2009) and divided redband trout abundance by two to account for both sexes. Estimates of redband trout spawner abundance at each site were then extrapolated for each GMU and subpopulation using the abundance estimation formula of Scheaffer et al. 1996.

N_e was approximated for subpopulations by multiplying the estimated number of mature adults in subpopulations by the range of N_e/N_{Adult} ratios derived via simulation above (Meyer et al. 2006).

Results

Estimation of Trout Abundance

We estimated there were nearly a million (925,069 ± 154,816) redband trout residing in the six GMUs during the study period (Table 2). The total study area abundance estimate for trout ≥ 100 mm TL (527,731 ± 95,846) was more precise than that for trout <100 mm TL (397,338 ± 121,579).

We concluded there were 34 redband trout subpopulations in the study area. Total abundance estimates (fish ≥ 100 mm) were obtained for all nine subpopulations believed to be small and isolated prior to initiation of the study and sampled more intensively relative to the standard EMAP sample draw. Estimates of redband trout in those streams ranged from 94 fish in Clover Creek to 16,965 in Sinker Creek (See Schill 2009, Table 5.4 for individual stream estimates).

We were less effective in obtaining total abundance estimates for the remaining 26 subpopulations using the sampling intensity level set for the EMAP sample draw. Few estimates of abundance within subpopulation reaches ranked *a priori* as "unknown" were calculated due to insufficient sample sizes. Total subpopulation estimates (fish ≥ 100 mm TL) were made for six of these subpopulations ranging in size from 15,145 to 101,333 fish (See Schill 2009, Table 5.5 for individual stream estimates).

Table 2. Estimates of total trout abundance (N_{census}) by size class, *a priori* distribution category, and estimated abundance of mature trout (N_{Adult}) for redband trout residing in six geographical management units (GMUs) in desert river tributaries to the Snake River of southwest Idaho, 1999-2005.

	≥ 100 mm TL				< 100 mm TL					
	Present		Unknown		Present		Unknown		Total abundance by GMU	Number of adult trout
GMU	N_{census}	± 90% CI	N_{census}	± 90% CI	N_{census}	± 90% CI	N_{census}	± 90% CI		N_{Adult}
Bennett Mtn	21,771	13,051	9,757	7,990	11,076	8,036	2,886	3,311	45,490	18,564
Bruneau R	110,692	36,956	6,295	6,216	52,689	23,149	47,018	72,789	216,694	52,657
North Front	49,536	32,192	35,960	27,191	40,568	43,740	62,168	58,094	188,232	35,425
Owyhee R	141,661	43,669	6,003	6,444	119,664	53,530	10,464	15,969	277,792	84,417
Rock Ck	17,579	14,356	84,864	59,701	8,715	7,711	21,050	15,939	132,208	40,061
Salmon Falls Ck	38,163	15,811	5,450	6,935	13,117	10,657	7,923	11,969	64,653	15,491
Total	379,402	156,035	148,329	114,477	245,829	146,823	151,509	178,071	925,069	246,615

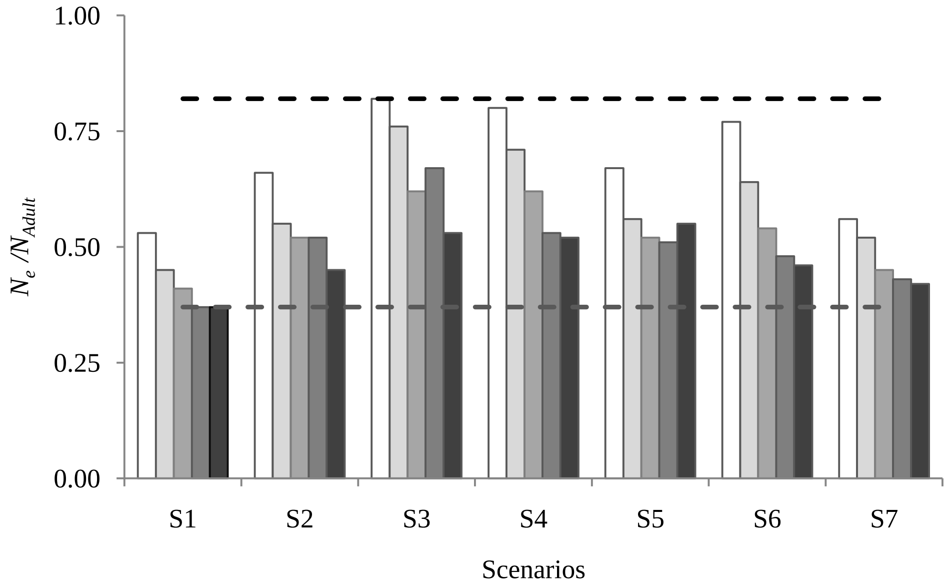


Figure 2. Ratios of estimated effective population size (N_e) to the mean number of adults, in 500 simulations of redband trout populations with varied life history patterns. The different bars within life history scenarios represent simulations for 200 years with mean adult number ranging from about 50 (leftmost bar) to 550 (rightmost bar). Dotted lines depict range of both N_e/N ratios observed (see text).

Estimation of N_e/N_{Adult} Ratios

The ratio of effective population size (N_e) to mean adult number (N_{Adult}) in the hypothetical populations ranged from 0.37 to 0.82 over the 200-year simulation period (Figure 2). Rounding the associated upper and lower bounds results in an N_e/N_{Adult} ratio of 0.4-0.8, values used in the estimation of subpopulation effective sizes reported below.

Estimation of Mature Redband Trout Abundance and Approximation of N_e

Within the 6 GMUs we estimated there were a total of 247,000 mature redband trout adults. Mature trout comprised, on average, 47% of the redband trout ≥ 100 mm TL of all redband trout estimated to reside in the study area. We approximated N_e in 17 of 34 subpopulations (50%) and obtained values ranging from 38 to about 48,000 (Table 3). The approximated range of all but three N_e estimates exceeded 1,000.

Discussion

Redband Trout Abundance

This study is the first field attempt to systematically identify the abundance and distribution of redband trout across arid basins at the upstream and southernmost margins of their Snake River range. The stream channel at nearly one-half (47%) of the randomly selected sites visited in this study was dry or nearly dry with no flow or trout present (n = 278). Our decision not to expend much sampling effort in “likely absent” stream reaches was justified based on the low occupancy rate of sampling sites in this stratum (just 3 of 139 sites).

Our study goal was not to attempt quantification of the amount of potential desert redband trout population range inhabited and lost (e.g. Thurow et al. 1997). Indeed, as High et al. (2008) noted, the true historical range of salmonids in such habitats is “unknown and unknowable”, a description that aptly applies to redband trout in remote, arid ephemeral drainages. As in Meyer et al. (2006), while working in the field, we observed a number of sizeable desiccated gulches that clearly had not contained flowing water for long periods, in some cases perhaps millennia (e.g. Poison "Creek") but were identified on the 1:100,000 GIS layer (and Thurow et al. 1997) as streams. Instead of speculating about where redband trout used to reside, we chose to

Table 3. Approximate range of N_e for redband trout in individual subpopulations by geographic management unit (GMU). Estimates for starred streams include all stream kilometer, streams not starred have estimates for *a priori* "likely present" stream kilometer only.

GMU	SubPopulation	Approximate range of N_e		
Bennett Mtn	Bennett Ck*	142	-	285
	Canyon Ck*	3,592	-	7,184
	Clover Ck*	38	-	75
	Cold Springs Ck*	1,045	-	2,090
	King Hill Ck*	1,250	-	2,501
	Little Canyon Ck*	1,301	-	2,602
Bruneau	Big Jacks Ck*	2,030	-	4,060
	Bruneau R	1,232	-	2,465
	Crab Ck			
	EF Bruneau R			
	Little Jacks Ck*	10,712	-	21,423
	Marys Ck			
	Sheep Ck			
North Front	Castle Ck	7,123	-	14,246
	Reynolds Ck*	1,824	-	3,647
	Shoofly Ck			
	Sinker Ck*	2,461	-	4,922
	Succor Ck			
Owyhee	Big Springs Ck			
	Cow Ck			
	Jordan Ck			
	NF Owyhee R	23,743	-	47,486
	SF Owyhee R	6,516	-	13,033
	Upper Owyhee R	3,348	-	6,696
Rock	Cedar Draw			
	Clear Ck			
	McMullen Ck			
	Rock Ck			
Salmon Falls	Cedar Ck			
	Lower Salmon Falls Ck			
	Shoshone Ck	389	-	778
	Trout Ck			
	Upper Salmon Falls Ck	4,798	-	9,597
	Willow Ck			

focus on where populations remain, along with their current size.

In the latter regard, it has been noted that "whatever aspect of ecology is being emphasized, the first essential for good work is to know how many animals there are in the area you are studying" (Andrewartha 1961). Despite the conservation biology focus on meta-population dynamics and patch size in risk assessments (Fritz 1979; Dunham and Rieman 1999), we believe the Andrewartha perspective is still valid and have attempted to quantify several important abundance metrics including total population sizes, numbers of mature adults, and N_e across the study area, both at the GMU and subpopulation scale. Focusing on density and population abundance metrics can allow general conclusions about current population or species status and risk to be made in the absence of detailed information on underlying processes affecting abundance such as habitat quality and interaction with other species (McElhaney et al. 2000).

Our extrapolations of population abundance, whether at the GMU or subpopulation scale should be viewed as conservative due to the negative sampling bias associated with both of the fish sampling techniques employed (Meyer et al. 2006; Schill 2009). Based on past studies, the true estimate of redband trout abundance across the entire study area likely far exceeds the 925,000 reported here. Although correction of such biases are possible, we have foregone such corrections, so as to present conservative estimates of abundance given uncertainty regarding other aspects of the study, such as low sample sizes in some stream order strata and the subjective nature of subpopulation boundary definition.

Past stocking of hatchery rainbow trout of coastal origin hinder the present study in terms of redband trout abundance and distribution. Preliminary findings of the companion genetics study suggest that 5 of 32 stream sites sampled contained hybridized populations (*O. mykiss irideus*), while the remaining 27 showed no evidence of introgression (M. Campbell, IDFG, unpublished data). Application of these results to the many genetically un-sampled streams across the landscape is difficult. Our plan to construct a simple model to examine the effects of stocking intensity and duration on the likelihood of eventual hybridization in the 32 streams genetically sampled to date has been hindered by problems merging several IDFG electronic stocking databases.

Until such issues are resolved, we are unable to make final estimates of non-hybridized redband trout abundance across the landscape.

N_e/N_{Adult} Ratio Simulations

Comparison of N_e/N_{Adult} ratios derived in this study with past efforts is difficult because of the lack of consistency in a number of factors considered in their derivation such as family size, fluctuating population size, and sex ratio (Frankham 1995). However, our truncated N_e/N_{Adult} ratio range (0.4-0.7) is only slightly lower than that recommended by Rieman and Allendorf (2001) for bull trout at 0.5-1.0, despite some sizeable differences in life history inputs, particularly longevity, age at maturity, and sex ratios.

Estimation of Mature Redband Trout Abundance and N_e estimates

The subpopulation estimates of mature adults and N_e reported in this study are "approximations based on approximations" (Rieman and Allendorf 2001). However, as Harris and Allendorf (1989) noted, for management purposes, it is probably unnecessary to strive for great precision in N_e estimation. Instead, these authors suggested relying on such approximations to assess relative risk among populations.

Assuming results of our subpopulation boundaries are reasonable, current genetic risk in terms of inbreeding or genetic drift for most redband trout populations in the study area appear to be low based on the 50:500 rule of thumb. Additional genetic sampling should be conducted in the 32 sites where genetic data has already been collected to obtain genetic estimates of N_e for direct comparison with our results.

Redband Trout Status

There are no simple answers to the question of what constitutes viable populations over the long term (Mace and Lande 1991). Over the past several decades, Population Viability Analysis (PVA) has emerged as one preferred method of assessing risk. Despite the proliferation of such studies, one of the earliest proponents of PVA analysis recently noted that such analyses have yet to successfully predict an extinction event that actually occurred, and has suggested that applying simple rules of thumb may be a superior and more useful approach for risk assessment in many instances (Shaffer et al. 2002). Based on a generalized simulation approach, Thomas (1990) suggested a mean of 5,500 individuals as a rough target for preserving a single isolated vertebrate population and observed that this general guideline is "reassuringly close to Soulé's low thousands" rule-of-thumb which was based on both empirical observations and conservation theory. Kautz and Cox (2001) concluded that maintaining 10 populations of at least 200 breeding adults would represent a sound conservation strategy for 11 key Florida species and ensure persistence over a 200-year period. Lastly, the 50/500 rule for N_e has been used extensively, particularly in the salmonid literature.

While we are unable to make final conclusions regarding redband trout population status across the arid southwest Idaho landscape until a final assessment of potential genetic effects is completed, existing results, in terms of population abundance estimates, number of subpopulations, and N_e estimates suggest that existing desert redband trout populations are not at appreciable risk in the near term when evaluated by the above rules of thumb.

Acknowledgements

T. Olsen and P. Larsen assisted with invaluable sampling design and site location. C. Kozfkay and M. Campbell generously provided genetic data, preliminary estimates of introgression and advice on population structuring. B. Zoellick assisted with *a priori* ranking of redband trout distribution and fieldwork. B, Rieman and B. Ardren provided advice on N_e/N_{Census} ratio estimation. Thanks to S. Elle, J. Cassinelli and a host of IDFG temporaries for conducting much of the field sampling for this effort. Special thanks to T. Lamansky and K. Meyer for sharing abundance data from the Salmonid Assessment Project in the study area and assistance with various phases of data analysis. L. Mamer assisted with GIS support, a remarkable amount of data management, and was a constant source of encouragement in this project from start to finish. The states of Oregon and Nevada and numerous private Idaho landowners graciously allowed us to sample trout within their borders or property. This project was funded by the Sport Fish Restoration Act and the Bonneville Power Administration.

Literature Cited

Allen, D. B., B. J. Flatter, and K. Fite. 1995. Redband trout (*Oncorhynchus mykiss gairdneri*) population and habitat surveys in Jump, Reynolds, and Sheep Creeks, and sections of the Owyhee River in Owyhee County, Idaho. Idaho Bureau of Land Management, Technical Bulletin 95-6.

Allendorf, F. W., D. Bayles, D. L. Bottom, K. P. Currens, C. A Frissell, D. Hankin, J. A. Lichatowich, W. Nehlsen, P. C. Trotter, and T. H. Williams. 1997. Prioritizing Pacific salmon stocks for conservation. Conservation Biology 11(1):140-152.

Andrewartha, H. G. 1961. Introduction to the study of animal populations. Methuen and Company, Ltd, London. 281 pages.

Ardren, W. R. and A. R. Kapuscinski. 2003. Demographic and genetic estimates of effective population size (N_e) reveals genetic compensation in steelhead trout. Molecular Ecology 12: 35-49.

Dunham, J. B. and B. E. Rieman. 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. Ecological Applications 9:642-655.

Frankham, R. 1995. Effective population size/adult population size ratios in wildlife: a review. Genetics Research (Cambridge) 66:95-107.

Fritz, R.S. 1979. Consequences of insular population structure: distribution and extinction of spruce grouse populations. Oecologia 42: 57-65.

Garant, D., J. J. Dodson, and L. Bernatchez. 2001. A genetic evaluation of mating system and determinants of individual reproductive success in Atlantic salmon (*Salmo salar* L.). The Journal of Heredity 92:137-145.

Harris, R. B., and F. W. Allendorf. 1989. Genetically effective populations size of large mammals: an assessment of estimators. Conservation Biology 3: 181-191.

High, B., K. A. Meyer, D. J. Schill, and E. R. J. Mamer. 2008. Distribution, abundance, and populations trends of bull trout in Idaho. North American Journal of Fisheries Management 28: 1687–1701.

Kautz, R. S. and J. A. Cox. 2001. Strategic habitats for biodiversity conservation in Florida. Conservation Biology 15:55-77.

Lacy, R. C., M. Borbat, and J. P. Pollack. 2005. VORTEX: a stochastic simulation of the extinction process. Version 9.50. Brookfield, IL: Chicago Zoological Society.

Mace, G. M. and R. Lande. 1991. Assessing extinction threats: toward a reevaluation of IUCN threatened species categories. Conservation Biology 5:148-157.

McElhaney, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionary significant units. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-NWFSC-42, 156 pages.

Meyer, K. A., D. J. Schill, J. A. Lamansky, Jr., M. R. Campbell, and C. C. Kozfkay. 2006. Status of Yellowstone cutthroat trout in Idaho. Transactions of the American Fisheries Society 135:1329-1347.

Miller, P.S. and R. C. Lacy 2005. VORTEX: a stochastic simulation of the extinction process. Version 9.50 User's manual. Apple Valley, MN. Conservation Breeding Specialists Group (SSC/IUCN).

Palm, Daniel. 2007. Restoration of streams used for timber floating: egg to fry survival, fry displacement, over-wintering and population density of juvenile brown trout. Doctoral thesis. Swedish University of Agricultural Sciences, Umeå. 2007:120.

Rieman, B. F. and F. W. Allendorf. 2001. Effective population size and genetic conservation criteria for bull trout. North American Journal of Fisheries Management 21: 756-764.

Scheaffer, R. L., W. Mendenhall III, and L. Ott. 1996. Elementary Survey Sampling, Fifth Edition. Duxbury Press, San Francisco.

Schill, D. J. and J. S. Griffith. 1984. Use of underwater observations to estimate cutthroat trout abundance in the Yellowstone River. North American Journal of Fisheries Management 4:479-487

Schill, D. J. 2009. Population studies of desert redband trout. Doctoral Dissertation. University of Idaho, Moscow, ID. 180pp.

Shaffer, M., L. H. Watchman, W. J. Snape III, and I. K. Latchis. 2002. Population viability analysis and conservation policy. *In* S. R. Beissinger and D. R. McCullough, editors. Population Viability Analysis. University of Chicago Press, Chicago, IL. Pages 123-142.

Stevens, D. L., and A. R. Olsen. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association 99: 262-278.

Thériault, V., L. Bernatchez, and J. J. Dodson. 2007a. Mating system and individual reproductive success of sympatric anadromous and resident brook char. *Salvelinus fontinalis*, under natural conditions. Behavioural Ecology and Sociobiology, 62: 51-65.

Thériault, V., D. Garant, L. Bernatchez, and J. J. Dodson. 2007b. Heritability of life-history and genetic correlation with body size in a natural population of brook char (*Salvelinus fontinalis*). Journal of Evolutionary Biology 20 (2007):2266-2277.

Thomas, C. D. 1990. What do real population dynamics tell us about minimum viable population sizes? Conservation. Biology 4: 324-327.

Thurow, R. F. 1987. Evaluation of the South Fork Salmon River steelhead trout fishery restoration program. Completion Report, US Department of the Interior, USFWS Lower Snake River Fish and Wildlife Compensation Plan. Contract 14-16-00001-86505, 1 March 1984 – 28 February 1986. Boise, ID.

Thurow, R. F., D. C. Lee, and B. E. Rieman. 1997. Distribution and status of seven native salmonids in the interior Columbia River Basin and portions of the Klamath River and Great basins. North American Journal of Fisheries Management 17:1094-1110.

Wright, S. 1969. Evolution and the genetics of populations, volume 2. The theory of gene frequencies. University of Chicago Press, Chicago, IL.

CONSERVING NATIVE TROUT AT THE LANDSCAPE SCALE USING THE WILD AND SCENIC RIVERS ACT

Scott Bosse
American Rivers, Northern Rockies Office, P.O. Box 1330, Bozeman, Montana 59771, USA

ABSTRACT — Signed into law by President Lyndon Johnson in 1968, the Wild and Scenic Rivers Act was passed at the height of the modern dam building era to protect some of the nation's most outstanding rivers in their free-flowing condition. Currently, the National Wild and Scenic Rivers System permanently protects more than 250 streams and 12,500 stream miles from new water development projects and other threats to water quality and water quantity. Historically, the Act has been employed mainly as a defensive tool to prevent the construction of new, federally licensed dams. However, beginning with the passage of the Craig Thomas Snake Headwaters Legacy Act in 2009, the Wild and Scenic Rivers Act was used for the explicit purpose of creating a *de facto* native fish refuge at the watershed scale. With climate change now affecting western rivers by altering hydrologic and thermal regimes, thereby increasing the demand for new water storage projects, it is no longer feasible to protect native trout *Oncorhynchus spp*. one stream at a time. Due to the unprecedented magnitude of this relatively recent threat, it is imperative that native trout be conserved at the landscape scale. As the nation's most powerful river protection law, the Wild and Scenic Rivers Act may be one of the best hopes for conserving wild trout in the 21st century.

INTRODUCTION

Native trout *Oncorhynchus spp.* across the interior West have experienced sharp declines over the last century due to a combination of factors including habitat degradation, dam construction, overfishing, and interactions with nonnative species (Dunham et al. 2003; Neville et al. 2006). Not only have the historic ranges of most native trout species contracted dramatically (Young 1995), but most remaining populations are fewer in number, exhibit less life history diversity, and are more fragmented, thus making them more prone to extirpation due to inbreeding and natural disturbances such as wildfires and floods (Clark et al. 2001; Poff et al. 2002; Westerling et al. 2006). Consequently, every native trout and char *Salvelinus spp*. species in the interior West either has been petitioned for listing under the federal Endangered Species Act, or, in the case of bull trout *S. confluentus*, already has been listed.

As if native trout in the interior West did not face enough challenges, climate change has exacerbated these problems by altering flow and thermal regimes and providing a competitive advantage to nonnative fish species that are better equipped to persist in a warmer environment (Rahel 2008). In the interior West, climate change is manifesting itself in the form of decreased snowpack, earlier runoff, lower summer stream flows, higher summer water temperatures, and increased frequency and severity of droughts, wildfires, and floods (IPPC 2007; Williams et al. 2009). Williams et al. (2009) predicted these factors will put as much as 73% of habitat currently occupied by Bonneville cutthroat trout *O.clarkii utah*; 65% of habitat occupied by westslope cutthroat trout *O. c. lewisi*; and 29% of habitat occupied by Colorado River cutthroat trout *O. c. pleuriticus* at high risk. Likewise, Rieman et al. (2007) concluded that projected temperature increases of 1C to 6C over the next 50 years could reduce coldwater habitat for bull trout by 18-92%.

While most state fish and game agencies and federal land management agencies in the West have done an admirable job developing plans to conserve native trout populations, implementation of these plans is constrained by limited funding, a lack of public support by anglers and non-anglers alike, and a strong tendency to take an incremental approach. Consequently, most native trout conservation is done at the local scale, one headwaters stream reach at a time, and it tends to be more focused on restoring diminished native trout populations than securing healthy ones. A typical native trout restoration project consists of installing an impassable barrier, applying piscicides to eradicate nonnative fish upstream of the barrier, and restocking the vacant habitat with native trout. Opportunities to link these isolated headwaters populations to create an interconnected metapopulation where genetic interchange can occur and diverse life histories can be expressed, however, are often missed. Given the unprecedented magnitude of the threats posed by habitat degradation, interactions with nonnative fish, and climate change, this incremental approach is inadequate as a 21st century native fish conservation strategy. What is needed today is an ambitious new approach to protect the last remaining native trout strongholds before they are compromised.

Native Fish Refuges

The concept of proactively protecting native salmonid strongholds is not new. Rahr et al. (1998) proposed creating salmon sanctuaries as a promising strategy for protecting native salmon in the 21st century, citing the creation of the nation's first and last salmon sanctuary on Afognak Island, Alaska in 1892. In making the case for the Afognak sanctuary at the 21st annual meeting of the American Fisheries Society, proponent Livingston Stone said:

> "Provide some refuge for the salmon, and provide it quickly, before complications arise which may make it impracticable, or at least very difficult….If we procrastinate and put off our rescuing mission too long, it may be too late to do any good. After the rivers are ruined and the salmon gone, they cannot be reclaimed….all the power of the United States cannot restore salmon to the rivers after the work of destruction has been completed" (Stone 1892).

Rahr et al. (1998) described three elements that had to be present to justify the creation of a salmon sanctuary: (1) sufficiently complex and connected habitats, and the biophysical processes to create and maintain those attributes through time; (2) native populations of Pacific salmon capable of expressing a major part of their historical phenotypic (or life history) diversity; and (3) adequate protection to ensure persistence through time. Ideally these salmon sanctuaries would be large enough to accommodate natural disturbances such as wildfires and floods. Among the watersheds that the authors identified as strong candidates for sanctuary status were the Bear River watershed in western Washington and the Trask River watershed in western Oregon. The authors envisioned these salmon sanctuaries being created by new state and federal designations.

The Wild and Scenic Rivers Act

The federal Wild and Scenic Rivers Act (16 U.S.C. 1271 et seq.) provides one such means of creating native trout refugia. Signed into law by President Lyndon Johnson in 1968, the Act was passed at the height of the modern dam building era in order to protect some of the nation's most outstanding rivers in their free-flowing condition. Currently the National Wild and Scenic Rivers System includes more than 200 streams and 12,500 stream miles (http://www.rivers.gov/publications/rivers-table.pdf). The idea for the Act was first proposed by brothers John and Frank Craighead in the mid 1950s (Palmer 1993). At the time, John Craighead was battling the U.S. Army Corps of Engineers to stop the Spruce Park Dam from being built on the Middle Fork of the Flathead River in northwest Montana. In an article that appeared in *Montana Wildlife* magazine, Craighead wrote, "Rivers and their watersheds are inseparable, and to maintain wild areas we must preserve the rivers that drain them"(Palmer 1993).

Section 1 of the Wild and Scenic Rivers Act describes its purpose and need:

> "It is hereby declared to be the policy of the United States that certain selected rivers of the Nation, which, with their immediate environments, possess outstandingly remarkable scenic, recreational, geologic, fish and wildlife, historic, cultural, or other similar values, shall be preserved in free-flowing condition, and that they and their immediate environments shall be protected for the benefit of present and future generations. The Congress declares that the established national policy of dam and other construction at appropriate sections of the rivers of the United States needs to be complemented by a policy that would preserve other selected rivers or sections thereof in their

> free-flowing condition to protect the water quality of such rivers and to fulfill other vital national conservation purposes" (16 U.S.C. 1271 et seq.).

The Wild and Scenic Rivers Act is often described as our nation's strongest federal river protection law. Its sharpest teeth are found in Section 7, which prohibits the Federal Energy Regulatory Commission (FERC) from licensing the construction of "any dam, water conduit, reservoir, powerhouse, transmission line, or other project works under the Federal Power Act" on or directly affecting any river which is designated as a component of the national wild and scenic rivers system. Section 7 further prohibits any department or agency of the United States from assisting by loan, grant, or license the construction of any water resources project that would have a direct and adverse effect on the values for which such river was established.

In addition to banning all federally-licensed dams and other harmful water development projects, the Wild and Scenic Rivers Act has the following mandates: (1) prohibits any federally assisted projects (e.g., highway construction, logging, mining, oil and gas drilling, etc.) that would impair a designated river's outstandingly remarkable values; (2) requires that water quality at the time of designation be maintained and enhanced; (3) creates a federally reserved water right for the minimum amount of flow necessary to sustain a designated river's outstandingly remarkable values; and (4) requires the development of a comprehensive river management plan to guide management of designated rivers for a 10-20 year period.

The original intent of the Wild and Scenic Rivers Act was clear – to prevent new dams from being built on designated rivers (Hiser 1988; Raffensperger and Tarlock, 1993). At the time the Act passed, an estimated 60,000-80,000 large dams blocked the nation's rivers, adversely affecting some 600,000 stream miles, or 17% of the nation's stream mileage (http://www.rivers.gov/waterfacts.html; Palmer 1993). It is probably safe to say that neither the Craighead brothers nor Idaho Senator Frank Church, the lead proponent of the Act in Congress, thought about using it proactively to protect the West's best remaining native trout strongholds. At the time, native trout and salmon runs were still relatively healthy by today's standards, and riparian degradation was not yet recognized as a serious threat (Palmer 1993).

In reviewing every piece of Wild and Scenic Rivers Act legislation that passed in its first 40 years of existence, this author could find only two references to protecting native fish. The 1980 Central Idaho Wilderness Act (16 U.S.C. 1132), which also granted wild and scenic status to a portion of the Salmon River, contained the following language in the findings section that presumably alluded to the river's native salmon and steelhead:

> "These wildlands and a segment of the Salmon River should be incorporated within the National Wilderness Preservation System and National Wild and Scenic Rivers System in order to provide statutory protection for the lands and waters and the wilderness-dependent wildlife and the resident and anadromous fish which thrive within this undisturbed ecosystem" (16 U.S.C. 1132)

Two decades later, the Steens Mountain Cooperative Management and Protection Act of 2000 (16 U.S.C. 460nnn) granted wild and scenic status to additional sections of the Donner und Blitzen River in southeast Oregon and at the same time designated the Donner und Blitzen Redband Trout Preserve. The stated purpose of the trout reserve was to

> "conserve, protect, and enhance the Donner und Blitzen River population of redband trout and the unique ecosystem of plants, fish and wildlife of a river system" (16 U.S.C. 460nnn-72)

Other than these two bills, no other legislation in the history of the Wild and Scenic Rivers Act specifically mentioned native fish until 2009.

The Snake Headwaters Model

The genesis of the Craig Thomas Snake Headwaters Legacy Act (16 U.S.C. 1271) was an aquatic conservation assessment of the Greater Yellowstone Ecosystem that identified the Snake Headwaters watershed in northwest Wyoming as the region's most intact native fish stronghold most in need of protection (Van Kirk 1999). Centered around Jackson Hole, the Snake Headwaters are home to abundant, genetically pure, interconnected popula-

tions of Yellowstone cutthroat trout (*O. c. bouvieri*) and Snake River finespotted cutthroat trout (*O. c. behnkei*). The only major hydrologic alteration in the watershed is the U.S. Bureau of Reclamation's Jackson Lake Dam, which originally was constructed at the outlet of Jackson Lake in 1906 to provide irrigation storage for eastern Idaho farmers.

Recognizing a unique opportunity to use the Wild and Scenic Rivers Act to proactively protect one of the West's last, best native trout strongholds while also shining a national spotlight on the watershed's outstanding recreational attributes, a coalition of conservationists, anglers and boaters launched the Campaign for the Snake Headwaters in 2003 with a goal of passing the first watershed scale wild and scenic legislation in the Act's history. The campaign reached a major milestone in May 2007 when Wyoming Senator Craig Thomas introduced a bill to protect 14 rivers and streams and 443 river miles in the Snake Headwaters watershed (Brenner 2007a). During its Senate subcommittee hearing, world-renowned fly fisherman Jack Dennis of Jackson, Wyoming explained why the bill was necessary:

> "I've been almost everywhere in the world there's rivers; this is one of the last great places left, and it needs to be protected" (Straub 2007).

The primary purpose of the bill was not to prevent the construction of dams, which was the original intent of the Wild and Scenic Rivers Act, but to create the very type of native fish sanctuary that Livingston Stone called for at the close of the 19th century.

One month after the bill was introduced, Senator Thomas died from leukemia. The man who was selected to replace him, current Wyoming Senator John Barrasso, eventually picked up the legislation, cut out some river miles in Lincoln County where the bill was controversial, and renamed it the Craig Thomas Snake Headwaters Legacy Act to honor its original sponsor.

In April 2009, after two years of legislative wrangling during which Idaho Senator Larry Craig unsuccessfully tried to eviscerate the bill out of fear it would adversely affect eastern Idaho irrigators (Brenner 2007b), the Craig Thomas Snake Headwaters Legacy Act finally passed as part of the Omnibus Public Lands Management Act of 2009 (16 U.S.C. 1). The final version of the bill protected 13 rivers and streams and approximately 400 stream miles, from the headwaters of the Snake River in Yellowstone National Park downstream to Palisades Reservoir on the Wyoming-Idaho border. The opening sentence in the findings section of the bill read:

> "The headwaters of the Snake River System in northwest Wyoming feature some of the cleanest sources of freshwater, healthiest native trout fisheries, and most intact rivers and streams in the lower 48 States" (16 USC 1271)

Because the Wild and Scenic Rivers Act was used to protect the Snake Headwaters, one of the West's most important native trout strongholds will forever remain free of dams and other harmful water development projects; the exceptional water quality of its rivers and streams will be safeguarded in perpetuity; and their flows will be maintained at sufficient levels to sustain native fish and wildlife. Moreover, the U.S. Forest Service and National Park Service, which together manage virtually all the public lands in the watershed, have begun work on a comprehensive river management plan for the Snake Headwaters that makes the conservation of native trout its highest priority (Bridger Teton National Forest 2010).

Discussion

Due to the diversity and magnitude of threats facing native trout and the difficulty of passing sweeping public lands legislation in a region that is leery of federal initiatives, there is no silver bullet when it comes to conserving native trout in the interior West. Turning the tide on native fish conservation will require employing a suite of strategies that includes habitat restoration, nonnative fish removal, water quality improvement, in-stream flow restoration, dam removals, as well as federal legislation aimed at protecting native fish strongholds where they still exist, be it through wilderness or wild and scenic rivers designation. The most important lesson to be learned from the Snake Headwaters model is that it represents the type of bold, innovative thinking that is needed to rescue native trout in the face of historic threats that have been exacerbated by climate change.

While the Craig Thomas Snake Headwaters Legacy Act serves as a model for how to protect native trout strongholds at a landscape scale, the Wild and Scenic Rivers Act clearly has its limitations: (1) it is still viewed as controversial in much of the interior West, although sweeping wild and scenic legislation recently passed in the ultra-conservative states of Wyoming, Idaho and Utah; (2) it is not a politically practical tool for rivers flowing primarily through private lands; (3) it only regulates land use within a half-mile wide corridor; (4) it does empower the federal government to restrict harvest levels or change outdated fish stocking policies; and (5) it addresses the symptoms of climate change (the demand for more water development projects), but not the cause (excessive carbon emissions).

Despite its weaknesses, the Wild and Scenic Rivers Act is a more practical and effective tool for protecting native trout than its terrestrial counterpart, the Wilderness Act. While the Wilderness Act has been used successfully to protect millions of acres of pristine land, most of it is higher in elevation and less biologically diverse than the riverside lands targeted by wild and scenic designation. As a former river specialist with the Department of the Interior once said, “One well-placed river with 50,000 acres might be more important than 500,000 acres of wilderness or national park on a glacier” (Palmer 1993). Another major advantage of the Wild and Scenic Rivers Act is that is can be used in watersheds that have experienced various levels of development, whereas the Wilderness Act by definition can only be used in large, intact, roadless watersheds.

Inspired by the success of the Snake Headwaters model but at the same time aware of its limitations, a broad coalition of river advocates recently launched a multi-year campaign to protect the best remaining free-flowing rivers and native trout fisheries in four major watersheds in western Montana. The campaign seeks to create a new model for river protection whereby the public lands reaches of selected rivers would be designated wild and scenic, and the lower reaches of the same rivers would be classified as coldwater conservation areas where federal funding could incentivize private landowners to boost in-stream flows, improve water quality, and protect ecologically vital riparian lands. Regardless of the outcome of this campaign, the effort demonstrates that the Wild and Scenic Rivers Act has evolved from being merely a means to stop new dams into an effective tool for proactively conserving native trout at the landscape scale.

References

Brenner, N. 2007a. Bill to protect Snake. Jackson Hole Daily. May 4, 2007.

Brenner, N. 2007b. Snake bill hits snag in Idaho. Jackson Hole News and Guide. August 3, 2007.

Bridger-Teton National Forest. 2010. Snake Headwaters wild and scenic comprehensive river management plan. Summer 2010 outreach publication.

Clark, M. E., K.A. Rose, D.A. Levine, and W.W. Hargrove. 2001. Predicting climate change effects on Appalachian trout: combining GIS and individual-based modeling. Ecological Applications 11:161-178.

Doppelt, R. 1993. The vital role of the scientific community in new river conservation strategies. Journal of the North American Benthological Society 12(2):189-193.

Dunham, J. B., M. K. Young, R. E. Gresswell, and B. E. Rieman. 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasion. Forest Ecology and Management 178:183-196.

Hiser, E. L., 1988. Piloting the preservation/development balance on wild and scenic rivers. Duke Law Journal 1988(1):1044-1079.

IPPC (Intergovernmental Panel on Climate Change). 2007. Climate change 2007: the physical science basis. Available: www.ipcc.ch. (February 2007).

Neville, H. M., J. B. Dunham, and M. M. Peacock. 2006. Landscape attributes and life history variability shape genetic structure of trout populations in a stream network. Landscape Ecology 21:901-916.

Palmer, T. 1993. The wild and scenic rivers of America. Island Press. Washington, D.C. and Covelo, CA.

Poff, N. L., M. M. Brinson, and J. W. Day. 2002. Aquatic ecosystems and global climate change: potential impacts on inland freshwater and coastal wetland ecosystems in the United States. Pew Center on Global Climate Change, Arlington, Virginia.

Raffensperger, C. and A. D. Tarlock. 1993. The Wild and Scenic Rivers Act at 25: The need for a new focus. Rivers 4(2): 81-90.

Rahel, F. ., B. Bierwagen and Y. Tanigughi. 2008. Managing aquatic species of conservation concern in the face of climate change and invasive species. Conservation Biology 22: 551-561.

Rahr, G.R., J. A. Lichatowich, R. Hubley, and S. M. Whidden. 1998. Sanctuaries for native salmon: A conservation strategy for the 21st century. Fisheries 23(4):6-7, 36.

Rieman, B.E. and D. Isaak. 2007. Anticipated climate warming effects on bull trout habitats and populations across the interior Columbia River Basin. Transactions of the American Fisheries Society 136:1552-1565.

Stone, L. 1892. A salmon national park. Transactions of the twenty-first annual meeting of the American Fisheries Society 21:149-162. Bethesda, Maryland.

Straub, N. 2007. River bill gets support: Famous angler testifies before senate subcommittee. Casper Star-Tribune. May 16, 2007.

Van Kirk, R., 1999. Status of Fisheries and Aquatic Habitats in the Greater Yellowstone Ecosystem. Greater Yellowstone Coalition, Bozeman, Montana.

Westerling, A.L., H.G. Hidalso, D.R. Cayan, and T.W. Swetnam. 2006. Warming and earlier spring increases western U.S. forest wildfire activity. Science 313:940-943.

Williams, J.E., A.L. Haak, H.M. Neville, and W.T. Colyer. 2009. Potential consequences of climate change to persistence of cutthroat trout populations. North American Journal of Fisheries Management 29:533-548.

Young, M.K., editor. 1995. Conservation assessment for inland cutthroat trout. U.S. Forest Service General Technical Report RM-GTR-256.

Use of Radio Telemetry with Mark-Recapture Snorkel Surveys to Improve Population Estimates of Fluvial Trout Species

James S. Baxter
Presenter; Fish and Wildlife Compensation Program-Columbia Basin, 103-333 Victoria Street, Nelson, British Columbia, CANADA, V1L 4K3, James.Baxter@bchydro.com

John Hagen
Corresponding Author; J. Hagen and Associates, 1663 5th Ave., Prince George, British Columbia, CANADA, V2L 3M2, hagen_john2@yahoo.ca

ABSTRACT — We used radio telemetry in conjunction with diver-based surveys to develop a predictive model for monitoring a population of rainbow trout *Oncorhynchus mykiss*. Specifically, we investigated the effect of variation in underwater visibility on the accuracy of diver counts over a 3-year period. Observer efficiency (number of tags seen relative to the number known to be present) was significantly related to horizontal underwater visibility in two of the three years of study. In the combined 3-year data set, horizontal visibility, which ranged from 5.0 to 16.5 m (average = 10.9 m), was a significant predictor of diver observer efficiency (r^2 = 0.62, P < 0.001), which ranged from a minimum of 0.13 to a maximum of 0.89 (average = 0.54). The use of radio tags allows the number of marks in a section and the observer efficiency of the surveyors to be directly estimated, which is not possible in traditional mark-recapture studies, and thus improves the population estimate. Ongoing population monitoring over many years suggests that there has been an increase in the abundance of trout associated with the implementation of a catch-and-release zone, and that our population estimates by diver counts can be considered reliable as a result of undertaking the observer efficiency study.

Introduction

Trout fisheries in the rivers and streams of the world provide social, environmental, recreational, and economic benefits to the communities around which they are based. As such, these fisheries require proper management to ensure their continued persistence and success. In general, over time the angler demand for quality trout stream fisheries has grown, but in some cases with the increase in angler effort there has not been an increase in population monitoring to address potential conservation, harvest, or management concerns. When coupled with the potential impacts of increased land development, water use, and hydroelectric operations, it is readily apparent that the monitoring of trout populations over time is of importance.

Diver-based underwater counting is one technique that is used for trout population monitoring where either the number of trout are directly estimated, or a population estimate is derived through mark-recapture, where the diver counts are the recapture technique. This method is typically employed as the surveys are relatively quick and less expensive to conduct than creel surveys. Mark-recapture studies employing diver counts as the recapture method have also been used to independently evaluate the accuracy of population estimates derived from diver counts, but from these it appears that system-to-system variability can be high, with species differences and the amount of in-stream cover being potential variables that can affect the accuracy of diver counts (Slaney and Martin 1987; Zubik and Fraley 1988; Young and Hayes 2001). Quantitative descriptions of factors that affect accuracy of diver counts within a given river system and salmonid population could allow the development of models for population estimation that account for changes in these variables from year-to-year, and within a season, thereby improving the precision of population estimates.

To evaluate the reliability of diver counts for assessing trout population status on several systems in southeastern British Columbia, we have conducted multi-year programs of radio telemetry research in conjunction with diver-based surveys. This has been done to understand the effects of changes in stream physical conditions such as underwater visibility and discharge on the accuracy of underwater counts of fluvial rainbow trout *Oncorhynchus mykiss* and westslope cutthroat trout *O. clarkii lewisi*. In general, divers obtained counts of trout in various size categories in stream sections in which radio-tagged fish that had also received a visual mark were present. We related the accuracy of the counts (number of tagged fish seen by divers as a proportion of the number known to be present – hereafter referred to as observer efficiency) to levels of horizontal underwater visibility and discharge on the survey dates. The goal of the research was to parameterize a model, if possible, for expanding future dive counts (without mark-recapture) into a population estimate and for estimating confidence intervals. This paper reports on the methods we have used to develop effective and reliable monitoring for a population of rainbow trout in one British Columbia river.

Methods

Our study was located on the Salmo River in southeastern British Columbia. The stream originates in the Selkirk Mountains 12 km southeast of the town of Nelson, and flows in a southerly direction for approximately 60 km from its origin to the confluence with the Pend d'Oreille River (Seven Mile Reservoir). The Salmo River is a 5th order stream and has a total drainage basin area of roughly 1,300 km^2.

We replicated underwater enumeration of rainbow trout of various size categories over a 3-year period in a 9-km section of the Salmo River mainstem under typical July water conditions. Prior to the July diver counts, radio tags were deployed in a sample of rainbow trout that had been captured by angling (see Hagen and Baxter 2005 for methods). Fish selected for tagging were a minimum of 35 cm fork length, although almost all radio tagged fish were longer than 40 cm. The latter length corresponded to the estimated minimum size of maturity for the population as determined from visual observations of maturity status and scale analysis, and therefore the population segment of greatest interest for conservation purposes. Pairs of T-anchor tags were also inserted into each fish's back on either side of and adjacent to the posterior insertion of the dorsal fin, ensuring that tagged fish could be identified from either side. Colours of T-anchor tags were unique for each year of the study. Only radio tags deployed during the same year, that each observer efficiency study was conducted in, were used for estimates of observer efficiency, so that the estimate of the number of tags still functioning in live fish could be considered reliable. Radio-tagged fish were identified by their orange (2001-tagged), white (2002-tagged), or blue (2003-tagged) anchor tags, and observations were noted for comparison with concurrent telemetry results from that survey date. Underwater horizontal visibility (horizontal secchi disk distance) was recorded three times during each diver survey, at the beginning and completion of the survey and once at midday.

For each year of the study, we described the relationship between observer efficiency and horizontal visibility using simple linear regression on arcsine square root-transformed data (Zar 1996). A comparison of observer efficiency relationships among years was conducted prior to pooling data for an overall regression analysis for the 3-year period, and was made using multiple analysis of covariance (Tabachnick and Fidell 2001). We also used regression analyses on untransformed data to describe the relationships between diver counts of untagged fish and horizontal visibility. Our two goals in the regression analyses were to conclude whether (1) diver observer efficiency and (2) counts of untagged trout were significantly related to underwater visibility levels. To estimate precision we used percent relative error, defined in Krebs (1999) as the half-width of the confidence interval ((upper CL - lower CL) / 2) of a specific prediction as a percentage of the prediction itself.

Rainbow trout population abundance monitoring in the entire main stem Salmo River, through the use of adjusted snorkeler counts, has continued in each year since the completion of the observer efficiency study in 2003. This has been done to monitor the response of the trout population to management changes (implementation of a catch-and-release section) and habitat enhancements. The population estimates for the surveyed length of the Salmo River main stem were calculated according to:

$$N = \sum_{i=1}^{k} C_i / \lambda_i$$

where N is the population estimate (30-40 cm or > 40 mm), C_i is the diver count for section i, λ_i is the estimated observer efficiency for section i (derived from the observer efficiency study), and k is the total number of stream sections. The overall population estimate, and associated estimates for kill and catch-and-release sections, was made by summing the point estimates for individual sections. We estimated confidence intervals, however, from the 2.5% and 97.5% percentiles of 1,000 estimates of N (for each of the two size distributions), where λ_i was simulated stochastically for each section based on the overall relationship of snorkeling efficiency to underwater visibility (Hagen and Baxter 2005). Standard errors for individual predictions of observer efficiency were computed using formulae in Zar (1996), and these formed the basis for the stochastic simulations.

Results

Eighteen, 10, and 10 radio transmitters were deployed in rainbow trout captured in our counting section during spring 2001, 2002, and 2003, respectively. For all 3 years of the study, the telemetry record indicated that most radio-tagged fish made only relatively small scale movements during the month of July. The average radio-tagged trout spent 89.3, 91.4, and 97.5% of the study period within the section during each of the three respective years.

Observer efficiency estimates made from observations of radio-tagged trout were positively related to levels of horizontal visibility in our study reach, but the strength of the predictive relationship was inconsistent among years. During 2001, variability in horizontal visibility explained only 3.8% of the variability in observer efficiency for that year, and the regression was not significant ($P = 0.675$). In contrast, observer efficiency was significantly related to horizontal visibility in both 2002 ($P = 0.013$) and 2003 ($P < 0.001$), with 74% and 99.7%, respectively, of the variability in observer efficiency being explained by visibility changes.

Because we were interested in whether a general relationship between the variables could be described, and in describing the uncertainty in a more realistic manner, we investigated whether combining the three years of data was feasible. We were not able to detect differences among the annual observer efficiency regressions statistically (MANCOVA; $P = 0.92$ for arcsine square root-transformed observer efficiency data), although it should be noted that the power to detect such differences was limited by the small sample size in each regression. However, a visual inspection of the combined data set (Figure 1) also indicated that the annual data overlapped, suggesting that pooling the data was reasonable. In the combined 3-year data set, horizontal visibility, which ranged from 5.0 to 16.5 m (average = 10.9 m), was a significant predictor of diver observer efficiency ($r^2 = 0.62$, $P < 0.001$), which ranged from a minimum of 0.13 to a maximum of 0.89 (average = 0.54).

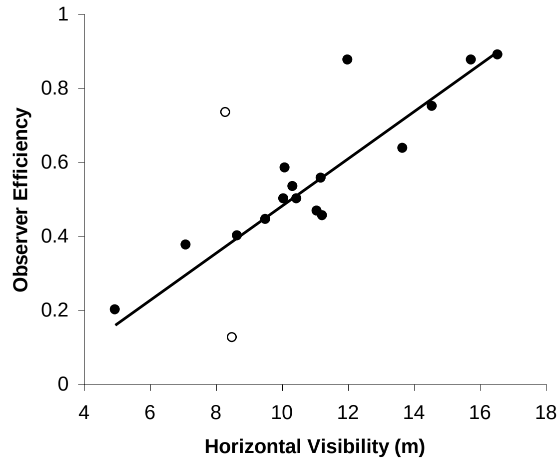


Figure 1. Observer efficiency estimates (closed circles: number of tagged fish seen relative to the number known to be present; open circles: outliers identified during pre-screening of data, but included in regression analysis) versus horizontal underwater visibility for three years' combined data from periodic diver surveys in the Salmo River, British Columbia.

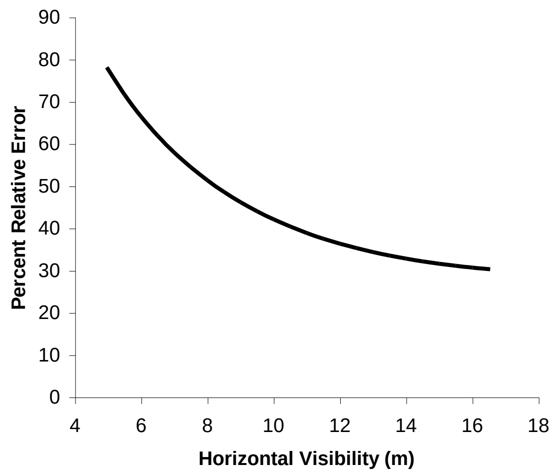


Figure 2. Percent relative error of specific predictions of observer efficiency versus horizontal visibility, estimated from the overall regression of arcsine square root-transformed observer efficiency on horizontal visibility for three years of combined data.

Plotted in terms of percent relative error (Figure 2), precision increased rapidly (i.e. percent relative error decreases) as horizontal visibility increased beyond 5 m, but the increase was much more gradual at higher levels of visibility. For all levels of horizontal visibility greater than 8.5 m, one-half confidence intervals for each observer efficiency prediction were within 50% of the value of the prediction itself, and for horizontal visibility greater than 13 m, one-half confidence intervals were within 35%.

To further investigate the relationship of observer efficiency to horizontal visibility, we also examined counts of untagged rainbow trout in the study reach (Figure 3). The notion that a positive, linear relationship exists between observer efficiency and horizontal visibility was supported by precise, significant relationships between counts of trout and visibility for both 2002 (>30 cm: r2 = 0.94, P <0.001; >40 cm: r2 = 0.93, P <0.001) and 2003 (>30 cm: r2 = 0.94, P = 0.007; >40 cm: r2 = 0.87, P = 0.021). The poor quality, nonsignificant relationships between counts of untagged trout and visibility in 2001 (>30 cm: r2 = 0.48, P = 0.083; >40 cm: r2 = 0.010, P = 0.85) were consistent with the relatively poor observer efficiency relationship for that year.

Five years of population estimates were calculated for evaluating the status of the Salmo River fluvial rainbow trout population (Table 1). Experimental introduction of a catch-and-release management zone occurred in 2003, but we did not expect to see population state changes reflected in

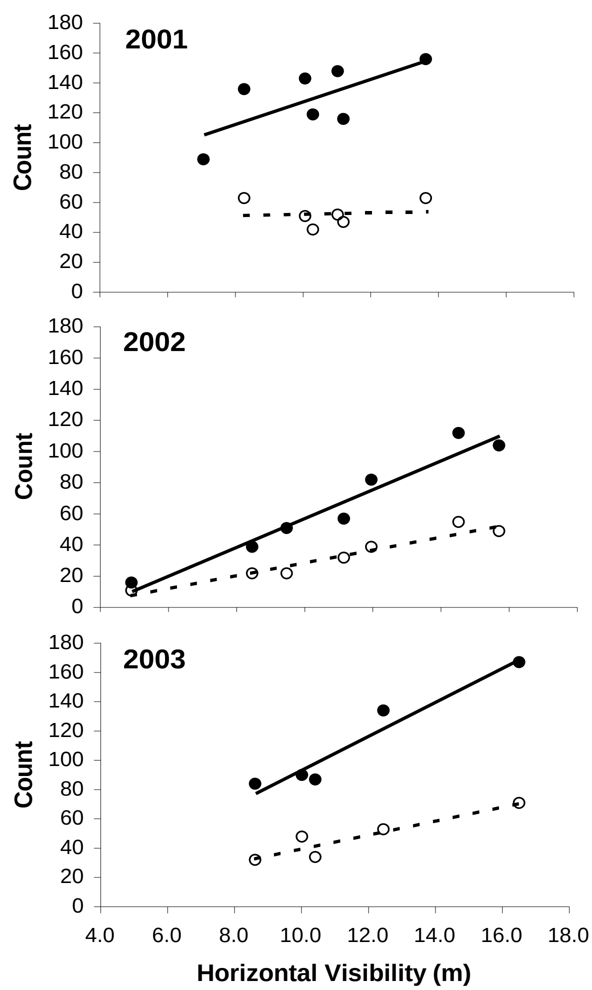


Figure 3. Counts of untagged rainbow trout >30 cm (solid circles: available for harvest) and >40 cm (open circles: adults) versus horizontal underwater visibility for 3 years of periodic surveys in the Salmo River, British Columbia.

Table 1. Population estimates (and confidence intervals) for rainbow trout 30-40 cm and >40 cm long in the Salmo River, British Columbia.

	Trout 30-40 cm			Trout >40 cm		
Year	Limited kill	Catch-and-release	Total	Limited kill	Catch-and-release	Total
2002	73(62-95)	144(119-202)	301(265-387)	56(48-72)	128(77-135)	174(152-228)
2003	80(70-102)	230(178-417)	504(441-738)	45(38-60)	120(92-216)	195(169-297)
2004	92(77-128)	196(159-286)	399(347-536)	70(58-97)	73(59-109)	165(145-217)
2005	246(209-397)	533(385-1037)	1082(884-1940)	89(75-144)	176(126-342)	306(248-536)
2006	332(288-429)	513(432-693)	1174(1051-1455)	72(62-96)	245(200-344)	366(314-484)

the 2003 population estimate, because the counts were conducted very early in the first angling season of the regulation change. Thus the effects of the catch-and-release regulation to date should be evaluated using the 2004-2006 population estimates for the limited kill and catch-and-release management zones (Table 1; Figures 4a, 4b).

Discussion

Any number of different factors may affect the observer efficiency (proportion of total number of fish present actually seen by divers) of underwater counts of fluvial salmonids by divers, including water clarity, pool depth, discharge, temperature,

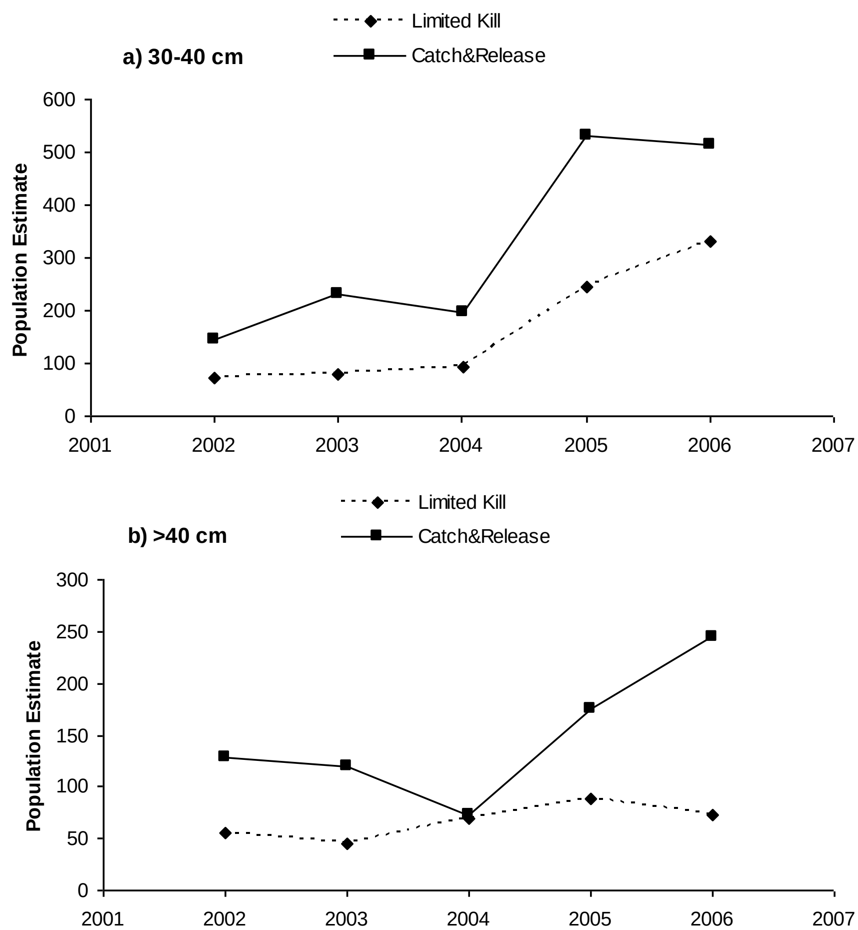


Figure 4. Population estimates in 2002-2006 for (a) 30-40 cm fork length and (b) >40 cm fork length (adult) rainbow trout in limited kill (one over 30 cm per day – diamonds and dashed line) and catch-and-release (squares and solid line) management zones of the Salmo River, British Columbia.

availability of cover, light intensity, and observer experience. In our study, variation in horizontal underwater visibility had a significant effect overall on the accuracy of diver counts of adult rainbow trout in our study reach.

With respect to fluvial salmonid populations, we are not aware of any studies that have investigated variability among years in factors that affect the accuracy of underwater counts. In our study the value of underwater visibility as a predictor of diver count accuracy remains unresolved because of ambiguous results from the first year's research. This result itself points to the value of longer time horizons for research into population assessment methods. Four or 5 years may represent a minimum level of commitment necessary to assess the reliability of methods across years.

There are only a small number of published accounts of the accuracy of underwater census techniques for fluvial salmonid populations, but from these studies it appears that system-to-system variability can be high (rainbow trout 59% at 7.6 m visibility: Northcote and Wilkie 1963; westslope cutthroat trout 74% at 3 m visibility: Slaney and Martin 1987; nonnative brown trout *Salmo trutta* 57-66% at 7 m visibility: Young and Hayes 2001). Differences in behaviour among the species studied in the above accounts may be a factor in variation among published observer efficiency estimates.

A principal drawback of the use of radio telemetry in mark-recapture studies is the costly nature of the equipment and the tags themselves, as well as the commitment involved in doing repeated counts, especially if done over a multi-year time period. However, outfitting marked fish with radio transmitters is attractive, because certain assumptions implicit in mark-recapture studies, such as (1) no emigration of marked individuals out of the study area, and (2) no mortality or harvest of tagged individuals, can usually be verified when the location of the radio transmitter is known with certainty. Marking fish without also deploying radio tags will likely be adequate if the marking events and re-sight swims are not greatly separated in time. The cost may be justifiable in instances where long-term population dynamics monitoring of a fish population is desired, and variability in the observer efficiency of divers is suspected to be an important component of sampling error.

The ongoing population monitoring over many years suggests that there has been an increase in the abundance of trout associated with the implementation of a catch-and-release zone, and that our population estimates by diver counts can be considered reliable. Further support for the idea that the catch-and-release management zone was having an effect on rainbow trout survival comes from a comparison of population sizes with the limited kill zone. For adult rainbow trout >40 cm in particular, population increases since the introduction of the regulation change have been principally restricted to the catch-and-release management zone. Interestingly, population increases in the 30-40 cm size class occurred in both zones. If these apparent increases accurately reflect the population state (the snorkeling efficiency study evaluated the accuracy of snorkeling counts of adult fish), they suggest an appealing alternative to total catch-and-release management, whereby migration from protected areas support a limited kill fishery while still maintaining a high effective population size to ensure a high probability of long-term conservation. A stronger conclusion from this experiment may have implications for design of regulations elsewhere, suggesting that continued monitoring of the population state is warranted. Our early results are consistent with the notion that a very high quality fishery is possible on the Salmo River, one that can meet the angling experience needs of a diverse cross-section of both harvest and catch-and-release anglers.

Acknowledgments

Funding for this project was provided by BC Hydro, the Columbia-Kootenay Fisheries Renewal Partnership, the Fish and Wildlife Compensation Program-Columbia Basin, Beaumont Timber, the BC Ministry of Environment, and the Salmo Watershed Streamkeepers Society. The successful completion of this study was made possible only as a result of the contributions of the following individuals: B. Green, L. Brazier, K. Andreashuk, R. Olmsted, G. Birch, D. Wilson, D. den Biesen, C. Tarala, J. Bell, M. Pearson, A. Chirico, C. Spence, J. Burrows, J. Hammond, B. Henderson, G. Nellestijn, K. Maloney, J. R. Baxter, S. Decker, A. Shepard, S. Arndt, H. Manson, G. Smith, D. Wassick, G. Corcoran, B. Tipple, A. Nellestijn, T. Baxter, Z. Baxter, and R. Roome.

References

Hagen, J., and J. S. Baxter. 2005. Accuracy of diver counts of fluvial rainbow trout relative to underwater visibility. North American Journal of Fisheries Management 25:1367-1377.

Krebs, C. J. 1999. Ecological Methodology. Benjamin/Cummings, Menlo Park, California.

Northcote, T. G., and D. W. Wilkie. 1963. Underwater census of stream fish populations. Transactions of the American Fisheries Society 92:146-151.

Slaney, P. A., and A. D. Martin. 1987. Accuracy of underwater census of trout populations in a large stream in British Columbia. North American Journal of Fisheries Management 7:117-122.

Tabachnick, B. G., and L. S. Fidell. 2001. Using Multivariate Statistics. Allyn and Bacon, Needham Heights, Massachusetts.

Young, R. G., and J. W. Hayes. 2001. Assessing the accuracy of drift-dive estimates of brown trout (*Salmo trutta*) abundance in two New Zealand Rivers: a mark-resighting study. New Zealand Journal of Marine and Freshwater Research 35:269-275.

Zar, J. H. 1996. Biostatistical Analysis. Prentice Hall, Upper Saddle River, New Jersey.

Zubik, R. J., and J. J. Fraley. 1988. Comparison of snorkel and mark-recapture estimates for trout populations in large streams. North American Journal of Fisheries Management 8:58-62.

ESA, WATER RIGHT ADJUDICATIONS, AND WESTERN SALMONID CONSERVATION: A TALE OF TWO SPECIES – BIG LOST RIVER MOUNTAIN WHITEFISH AND MONTANA FLUVIAL GRAYLING

Larry Zuckerman

Central Idaho Director, Western Watersheds Project, PO Box 1322, Salmon, Idaho 83467, (208) 756-8000, (208) 756-8008 FAX, larry@westernwatersheds.org

ABSTRACT — The Federal Endangered Species Act and adjudications of water rights by Western states are examined in the context of the conservation of Western salmonids. Case histories using the Big Lost River mountain whitefish *Prosopium williamsoni* and the Montana fluvial grayling Distinct Population Segment ("DPS") *Thymallus arcticus* are examined for the Big Lost River Basin of Central Idaho and the Big Hole River Basin of Southwestern Montana. An endemic fish in a closed basin, the Big Lost River mountain whitefish has a much more limited historic range than most of the Upper Missouri River Basin upstream of Great Falls, Montana, where Lewis and Clark first encountered Montana fluvial grayling. Yet, both fishes face similar threats and exhibit parallel declines. Limitations of ESA species listings, designations of critical habitats, section 7 consultations, and section 9 take prohibitions as well as voluntary conservation agreements and restoration projects are often magnified by constraints placed by Western States' water appropriation systems. The nexus of ESA and states' water rights may offer potential solutions for the conservation and recovery of declining Western salmonid species, especially in light of projected adverse effects to these High Desert, coldwater fish with impending climate change in the Interior West.

AMERICAN WHITE PELICAN PREDATION ON YELLOWSTONE CUTTHROAT TROUT IN THE BLACKFOOT RIVER SYSTEM, IDAHO.

David M. Teuscher
Idaho Department of Fish and Game, Pocatello, ID 83201, 208-232-4703, dteuscher@idfg.idaho.gov

Daniel J. Schill
Idaho Department of Fish and Game, Boise, ID 83714, 208-287-2777, dschill@idfg.idaho.gov

ABSTRACT — Growth of American White Pelican (AWPE) *Pelicanus erythrorhynchos* nesting colonies in southern Idaho since the early 1990s has generated concerns about the effect of pelican predation on Yellowstone cutthroat trout (YCT) *Oncorhynchus clarkii bouvieri* in the Blackfoot River System. Nesting AWPE on Blackfoot Reservoir has increased exponentially from 0 nesting birds in 1992 to 3,174 in 2009. The abundance of YCT migrating up the Blackfoot River from the reservoir to spawn declined from 4,747 in 2001 to a low of 16 in 2006, rebounding to 865 in 2009.

Documentation of pelican predation impacts on YCT began in 2002 and includes estimates of pelican exploitation on YCT, evaluation of bird scarring rates, and use of automated digital photography. To date, findings include two separate estimates of direct predation on migrating YCT using radio telemetry. The most recent, minimum estimate was completed in 2007, where 9 of 27 (33%) radio-tagged YCT were consumed by nesting AWPE. In 2004, 70% of the upriver migrating YCT were injured by birds as evidenced by fresh scars. Since 2007, remote cameras have been deployed along the Blackfoot River to record hourly use by AWPE. Over 40,000 digital images have been analyzed providing counts of peak pelican foraging numbers, changes in foraging behavior related to water conditions, and diel foraging patterns. Bird scarring rates and the digital photography results suggest that pelican predation impacts on migrating cutthroat trout are inversely proportional to river flows.

Predation results prompted the Idaho Department of Fish and Game to develop a pelican management plan. Implementation of the plan began in 2010, with the return of badgers and skunks to one of the pelican nesting islands on Blackfoot Reservoir. Those species were removed from the nesting islands in the early 1990s to enhance goose production. The presence of badgers may discourage pelican nesting on the island. A nesting exclusion fence was also tested in 2010. The exclusion fence covered about 50% of a second island used by nesting AWPE. Aerial photographs and ground surveys confirmed that no AWPE nested within the exclosure during the entire nesting season, which suggests that nonlethal methods may exist for controlling AWPE recruitment in the future.

INTRODUCTION

The impact of piscivorous birds on commercially and socially important fish stocks has been a broad concern throughout North America and Europe (Harris et al. 2008) and potential negative effects of AWPE populations on such fisheries are no exception (Lovvorn et al. 1999; Glahn and King 2004; King 2005). The number of AWPE in North America approximately doubled between 1980 and 2002, increasing by nearly 5% annually during that period (King and Anderson 2005). Keith (2005) reported North American AWPE populations increasing from 30,000 in 1933 to about 100,000 birds by 1985, to 400,000 birds by 1995, values that, when plotted, suggest continent-wide exponential population growth. While most of the continental AWPE population breeds east of the Continental Divide, numbers have also increased in many parts of the west (Findholt and Anderson 1995a) and in the western metapopulation collectively (King and Anderson 2005; Murphy 2005).

In southern Idaho growth of AWPE nesting colonies since the early 1990s has generated concerns about the effect of their predation on salmonids, especially on YCT in the Blackfoot Reservoir and upper Blackfoot River system (IDFG 2009). The native YCT stock stages for its annual spawning run at the mouth of the Blackfoot River, which lies only 8 km from Gull Island, the nearest AWPE nesting colony. Nesting AWPE on Blackfoot Reservoir have increased from 0 nesting birds in 1992 to 200 nesting birds recorded in 1993 to a peak in 2007 of 3,416 adult birds. Since 2001, the abundance of adfluvial YCT migrating up the Blackfoot River from the reservoir to spawn declined from 4,747 in 2001 to a low of 16 in 2006, rebounding to 540 fish in 2008 and 865 in 2009 (Brimmer et al. in review).

The potential for AWPE to consume biologically meaningful numbers of salmonids appears low, based on some diet studies. Pelicans require shallow water (typically 0.3-0.65 m) or fish that can be reached within 1.3 m of the surface of deep water (Anderson 1991; Ivey and Herziger 2006). In lentic circumstances, this leads to a diet predominantly comprised of nongame fish such as chubs *Gila* sp., suckers *Catostomus* sp., and common carp *Cyprinus carpio* (Knopf and Evans 2004; Teuscher 2004). On Pathfinder Reservoir in Wyoming, over 83% of the biomass consumed by AWPE was composed of white suckers *Catostomus commersonii*, common carp, and tiger salamanders *Ambystoma tigrinum* (Findholt and Anderson 1995a). At Chase Lake, North Dakota, tiger salamanders comprised the majority of prey items in terms of occurrence frequency and volume (Lingle and Sloan 1980).

However, AWPE are typically reported in the literature as highly adaptable, opportunistic foragers, readily selecting sites and prey that are most available (Hall 1925; Knopf and Kennedy 1980, 1981; Lingle and Sloan 1980; Flannery 1988; Findholt and Anderson 1995b), a trait that is problematic for some fish spawning aggregations. For example, AWPE seek out spawning concentrations of tui chub *Gila bicolor* at Pyramid Lake, particularly when they enter shallow littoral areas and display "quick jerking motions" associated with spawning (Knopf and Kennedy 1980). More recently, AWPE have been identified as a hindrance to conservation efforts for Cui-ui *Chasmistes cujus*, an ESA endangered adfluvial sucker that ascends the Truckee River from Pyramid Lake to spawn (Scoppettone and Rissler 2002). Because AWPE prey on adult Cui-ui immediately prior to spawning, their impact on this en-endangered species might be severe (Murphy 2005). Similarly, AWPE detect and use adfluvial YCT spawning aggregations in inlet rivers and streams. Davenport (1974) reported that adfluvial YCT were the preferred prey of AWPE in a study on Yellowstone Lake, an observation reiterated by Varley and Schullery (1996). In southeast Idaho, expanding AWPE are concentrating at the mouths of well known cutthroat trout spawning tributaries such as the Blackfoot River, and St. Charles and McCoy creeks (IDFG 2009). Documentation of the level of impact by AWPE predation on those cutthroat trout populations is lacking. In this study we estimate AWPE predation rates on migrating YCT in the Blackfoot River. Hourly foraging patterns and bird scars on YCT caused by AWPE are also reported.

Study Area

Blackfoot Reservoir is located in the southeast corner of Idaho at an elevation of 1,685 m (Figure 1). The reservoir covers 7,284 ha. The fish community is dominated by Utah chub, Utah sucker *Catastomus ardens*, yellow perch *Perca flavescens*, and common carp. Yellowstone cutthroat trout and hatchery-stocked rainbow trout *Oncorhynchus mykiss* make up less than 5% of the species composition in the reservoir. Rainbow trout and cutthroat trout have been stocked in the reservoir since its impoundment in 1912.

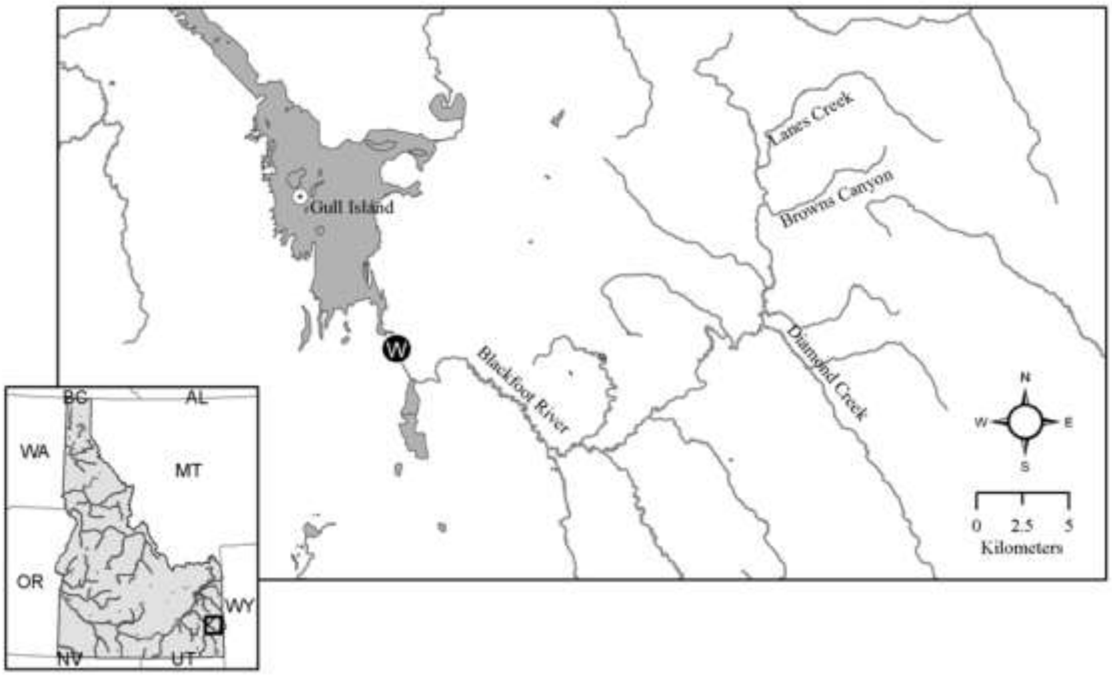


Figure 1. Study area showing the Blackfoot River, the Blackfoot Reservoir, Gull Island used by nesting AWPE, and the fish weir site (W) used for monitoring YCT escapement and collecting fish for implanting telemetry tags.

METHODS

In 2004 and 2007, radio transmitters were surgically implanted in YCT that were captured at the fish trap located on the Blackfoot River about 3.2 km upriver of the confluence with Blackfoot Reservoir (Figure 1). The radio transmitters were Advanced Telemetry Systems model F1300. Transmitters weighed about 11 g and were implanted in YCT ≥ 600 grams. The transmitter carried mortality signals that would deploy if held stationary for more than 24 h. The mortality indicators help determine mortality date, mortality location, and the potential cause of mortality (i.e., tags found in bird nests).

Surgery began by anesthetizing fish. Incisions were approximately 35 mm long, centered between the pectoral fins and pelvic fins. A grooved directional tool approximately 100 mm long was inserted into the incision and slid anteriorly, close to the flesh to prevent any contact with the internal organs. A 100-mm long catheter needle was inserted behind the pelvic fins and slid up the direction tool until it exits the opening of the incision. The antenna was inserted into the catheter needle and directed out the hole created behind the pelvic fins. The body of the tag was then inserted into the 35-mm incision. Incisions were closed with three or four stitches. Surgery times ranged from 6 to 8 min. Fish were placed in live wells filled with fresh water to recover and released about 100 m above the fish trap.

The AWPE predation rates were estimated by recovering fish telemetry tags from pelican nests on Gull Island. Several times during the nesting period, a boat was used to circle Gull Island. If a tag signal was received, the boat was docked and a person would recover the tag from the island. The predation rate estimate was made by dividing the number of telemetry tags recovered from pelican nests by the total number implanted in migrating YCT.

The IDFG monitors AWPE use of the Blackfoot River using automated digital cameras. Figure 2 shows an example of one camera location that has been in place since 2007. Additional camera sites have been deployed to monitor most of the lower 3.2 km of river. However, for purposes of this paper and because it provides the only continual data set from 2007, only data collected from camera location three is analyzed for trend use. The digital images (taken hourly) provide estimates of overall AWPE use, diel foraging patterns, and seasonal changes in use.

To monitor the adfluvial YCT population trends, IDFG operates a spawning migration trap on the Blackfoot River (Figure 1). In addition to counting and passing YCT upriver to spawn, the trap has been used as the collection site to radio tag YCT and evaluate fish for bird scars. Since 2004, all YCT caught in the trap have been visually inspected for bird scars (Figure 3). A fish was determined to have a bird scar if it had a puncture hole or deep slash marks occurring on both sides of its body. The matching wounds criteria ensured that the marks were made by an AWPE attempting to hold the fish with its bill.

RESULTS

In 2004, we tagged 28 YCT collected at the Blackfoot River weir and four (14%) were recovered from AWPE nests on Gull Island. In a repeat study completed in 2007, 9 out of the 27 (33%) fish tags were recovered from AWPE nests. The two-fold increase in predation was similar to the relative increase in the AWPE population (Table 1). The nesting colony increased from 1,748 in 2004 to 3,416 in 2007.

Figure 2. Digital images taken June 2007 that show AWPE use on the Blackfoot River. This camera location has been monitoring use of AWPE since 2007.

Figure 3 Birds scars on a YCT collected at the fish weir during its upriver spawning migration on the Blackfoot River.

Over 40,000 digital images have been reviewed to document AWPE use on the Blackfoot River. The images show that use on the river by AWPE begins as Utah suckers and YCT enter the river to begin their spawning migrations. Use continues through June and tapers off during the month of July. Changes in river flow also impact AWPE use. For example, over 5,340 AWPE were counted at one camera location during a 19-day period in 2007. At the same site, only 751 AWPE were counted during the same period in 2009. Average May river flows were 115 cfs in 2007 compared to 568 cfs in 2009 (Table 1). It was apparent from the digital images that many of the river rocks used by AWPE as foraging platforms in 2007 were under water in 2009.

Foraging patterns of AWPE during the day also varied by year. In 2007, AWPE use on the river peaked in the early morning and evening hours. In 2009, bird numbers were lowest in the morning and steadily increased throughout the day and peaked at 1800 hours (Figure 4). In 2008 – 2009, the cameras

Table 1. Numbers of YCT passed through the weir, AWPE nesting population, bird scarring rates, and average May discharge in the Blackfoot River. Bird scars were observed on YCT in 2003, but no daily scar records were kept.

Year	YCT escapement	AWPE nesting population	YCT with bird scars	Average May river discharge (cfs)
2001	4,747		None observed	74
2002	902	1,352	0%	132
2003	427	1,674	Observed	151
2004	125	1,748	70%	127
2005	16	2,800	6%	389
2006	19	2,548	38%	453
2007	98	3,416	15%	115
2008	548	2,390	10%	409
2009	865	3,174	14%	568

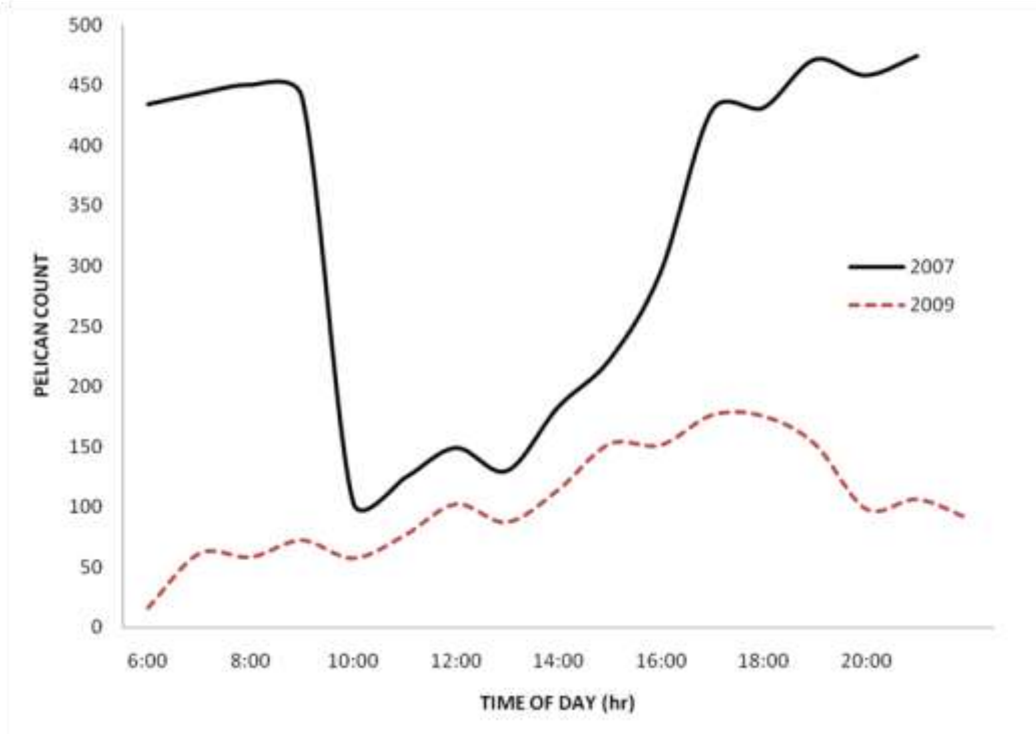


Figure 4. Hourly foraging patterns of AWPE at one camera location on the Blackfoot River. The counts for each hour were summed over a 19-d period from May 26 to June 13.

were set to record digital images at night. Nighttime foraging by AWPEs was observed, but the field of vision within the images was only a few meters. Because of the limited field of vision, AWPE counts from night images were not compared to the daytime counts.

Based on bird scarring trends on migrating YCT, an intense foraging pattern by AWPE developed on the Blackfoot River over a 4-year drought period (2001 to 2004). In 2001, trap tenders monitoring YCT populations observed no bird scars on YCT (Table 1). In 2002, 125 YCT randomly selected from the adfluvial spawning run were photographed. None of the 125 fish had bird scars. It was during the 3rd consecutive low water year (2003) that trap tenders began noticing bird scars on YCT. In 2004, 70% of the YCT collected at the weir suffered from bird scars.

Discussion

There are several limitations to this predation rate study. First, the estimates do not account for total predation, because AWPE likely excrete tags in other locations besides Gull Island. For example, in 2004, about 30% of recovered tags was found on lands other than Gull Island. Unidentified predators may have carried some of those tags out of the river corridor, but it is not possible from our dataset to determine how many were taken by AWPE. Secondly, most AWPE foraging activity observed in the digital images occurred below the fish weir. We captured and tagged YCT at the weir and predation below the weir was not measured; thus we believe the 2004 and 2007 predation rate estimates (14 % and 33%) are much underestimated. Future studies should focus on tagging YCT in the reservoir prior to spring migrations to obtain a total spawning run predation rate. In addition to improving the adult predation estimates, downstream emigrating juvenile salmonids are vulnerable to piscivorous birds (White 1957; Ruggerone 1986; Kennedy and Greer 1988) and should be included in future evaluations.

The use of automated digital images can be a useful tool for monitor AWPE abundance and foraging patterns. The cameras provide a cost-effective assessment tool that measures instantaneous AWPE use of about 3 km of the Blackfoot River. It would be prohibitively time consuming and costly to complete the same level of assessment using field crews.

Some of the limitations of the automated photography include equipment failure and weather conditions that obstruct the camera lens (i.e., snow).

In response to the measured predation impacts, IDFG developed a management plan that recommended significant reductions in the AWPE population nesting at the Blackfoot Reservoir (IDFG 2009). Implementation of the plan began in 2010, with the return of badgers and skunks to one of the pelican nesting islands on Blackfoot Reservoir. Those species were removed from the nesting islands in the early 1990s to enhance goose production. The presence of badgers may discourage pelican nesting on the island. A nesting exclusion fence was also tested in 2010. The exclusion fence covered about 50% of a second island used by nesting AWPE. Aerial photographs and ground surveys confirmed that no AWPE nested within the exclosure during the entire nesting season, which suggests that nonlethal methods may exist for controlling AWPE recruitment in the future.

Literature Cited

Anderson, J. G. T. 1991. Foraging behavior of the American White Pelican *(Pelecanus* erythrorhynchos in western Nevada. Colonial Waterbirds, 14(2):166-172.

Brimmer, A., D. Teuscher, R. Hillyard and M Kreizenbeck. In review. Fishery Management Annual Report, Southeast Region. Idaho Dept of Fish and Game, Boise.

Davenport, M. B. 1974. Piscivorous avifauna on Yellowstone Lake, Yellowstone National Park. U.S. National Park Service, Yellowstone National Park, Wyoming.

Derby, C. E., and J. R. Lovvorn 1997a. Predation on fish by cormorants and pelicans in a cold-water river: a field and modeling study. Canadian Journal of Fisheries and Aquatic Science. 54: 1480–1493.

Findholt, S. L., and S. H. Anderson. 1995a. Diet and prey use patterns of the American white pelican (*Pelecanus erythrorhynchos*) nesting at Pathfinder Reservoir, Wyoming. Colonial Waterbirds 18:58-68.

Findholt, S. L., and S. H. Anderson. 1995b. Foraging areas and feeding habitat selection of American white pelicans (*Pelecanus erythrorhynchos*) nesting at Pathfinder Reservoir, Wyoming. Colonial Waterbirds 18:47-57.

Flannery, A. W. 1988. Foraging habitat of white pelicans on Great Salt Lake marshes. Masters Thesis, Utah State University, Logan.

Glahn, J. F., and D. T. King. 2004. Biology and culture of channel catfish. Wildlife Damage Management, Internet Center for USDA National Wildlife Research Center, Staff Publications, Lincoln, Nebraska.

Hall, E. R. 1925. Pelicans versus fishes in Pyramid Lake. The Condor 27:147–160.

Harris, C. M., J. R. Calladine, C. V. Wernham, and K. J. Park. 2008. Impacts of piscivorous birds on salmonid populations and game fisheries in Scotland: a review. Wildlife Biology 14:395-411.

Idaho Department of Fish and Game. 2009. Management of American White Pelicans in Idaho, Boise.

Idaho Department of Fish and Game. 2010. Idaho White Pelican 2009 collecting permit annual report, Blackfoot River and Reservoir, southeast Idaho. Idaho Department of Fish and Game, Boise.

Ivey, G.L., and C.P. Herziger. 2006. Intermountain West Waterbird Conservation Plan, Version 1.2. A plan associated with the Waterbird Conservation for the Americas Initiative. Portland, Oregon.

Keith, J. O. 2005. An overview of the American White Pelican. Waterbirds 28 (Special Publication 1):9-17.

Kennedy, G. J. A., and J. E. Greer. 1988. Predation by cormorants, *Phalacrocorax carbo* (L.), on the salmonid populations of an Irish river. Aquaculture and Fisheries Management 19:159-170.

King, D. T. 2005. Interactions between the American White Pelican and aquaculture in the Southeastern United States: an overview. Waterbirds 28 (Special Publication 1):83-86

King, D. T., and D. W. Anderson. 2005. Recent population status of the American White Pelican: a continental perspective. Waterbirds 28 (Special Publication 1):48-54.

Knopf, F. L., and J.L. Kennedy. 1980. Foraging sites of white pelicans nesting at Pyramid Lake, Nevada. Western Birds 11:175–180.

Knopf, F.L., and J.L. Kennedy. 1981. Differential predation by two species of piscivorous birds. Wilson Bulletin 93:554–556.

Knopf, F.L., and R.M. Evans. 2004. American white pelican (*Pelecanus erythrorhynchos*). The Birds of North America Online (A. Poole, Ed.) Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/057

Lingle, G. R., and N. R. Sloan. 1980. Food habits of white pelicans during 1976 and 1977 at Chase Lake National Wildlife Refuge, North Dakota. Wilson Bulletin 92:123–125.

Lovvorn, J. R., D. Yule, and C. E. Derby. 1999. Greater predation by double-crested cormorants on cutthroat versus rainbow trout fingerlings stocked in a Wyoming river. Canadian Journal of Zoology 77:1984–1990.

Murphy, E. C. 2005. Biology and conservation of the American White Pelican: current status and future challenges. Waterbirds 28 (Special Publication 1):107-112.

Ruggerone, G. T. 1986. Consumption of migrating juvenile salmonids by gulls foraging below a Columbia River dam. Transactions of the American Fisheries Society 115:736-742.

Scoppettone, G. G., and P. Rissler. 2002. The endangered Cui-ui: an overlooked prey of American White Pelicans at Pyramid Lake, Nevada. Pacific Seabirds 29:64.

Teuscher, D. 2004. Regional fishery management investigations. Job Performance Report. Report F-71-R-28. Idaho Department of Fish and Game, Boise.

Varley, J. D., and P. Schullery. 1996. Yellowstone lake and its cutthroat trout. Pages 49-73 i*n* Science and Ecosystem Management in the National Parks, W. L. Halvorson and G. E. Davis (eds.). University of Arizona Press. Tucson

White, H. C. 1957. Food and natural history of mergansers on salmon waters in the Maritime Provinces of Canada. Fisheries Research Board of Canada Bulletin No. 16.

ASSESSING THE CAPTURE EFFICIENCY OF BARBED VS. BARBLESS ARTIFICIAL FLIES FOR TROUT

Roger K. Bloom

Senior Fisheries Biologist, Heritage & Wild Trout Program, California Department Fish and Game, 1701 Nimbus Road, Rancho Cordova, California 95670, Phone (916) 464-6355, Fax (916) 464-6358, email rbloom@dfg.ca.gov

ABSTRACT — We examined the capture efficiency of artificial flies fished with barbed and barbless hooks in various waters throughout California. Waters were selected based on high catch per unit effort and representative trout species. Artificial flies were standardized by J-style hooks and three artificial fly types (dry, nymph, and streamer). In an effort to reduce bias, anglers were not told what hook treatment they were using and were not allowed to handle or visually inspect the flies during the course of the sampling period. Capture efficiency differed significantly between hook treatment ($P < .0001$) for all angler experience levels and fly types combined. Mean capture efficiency for anglers using barbed flies was 0.757 *versus* barbless flies at 0.626. Results from this study provide valuable information on the effects that barbless hook regulations may have on managing inland trout waters. Fisheries managers must weigh the biological justification for barbless regulations with the potential reduction in catch rates and associated angler satisfaction.

INTRODUCTION

Fisheries managers are tasked with developing and maintaining quality sport fishing opportunities while balancing the need to protect and monitor the aquatic resources. Using sport fishing regulations as a management tool can have a substantial effect on the existing fisheries and, if used appropriately, can enhance angling opportunities. Currently, California has various freshwater fishing regulations that require the use of barbless hooks on rivers and lakes. These regulations were likely proposed and adopted based on the perception that barbless hooks would decrease hooking mortality by reducing handling time, stress, and trauma.

Although previous studies and assessments have referenced quicker unhooking times when using barbless hooks (Knutson 1987; Barnhart 1990; Schill and Scarpella 1997; Schaffer and Hoffman 2002; Meka 2004), the effects from barbless hooks on post-release survival have shown mixed results (Hunsaker et al. 1970; Falk et al. 1974; Wydoski 1977; Mongillo 1984; Talyor and White 1992; DuBois and Dubielzig 2004). Faragher (2004) provided an overview of existing studies on hooking mortalities for trout. He distilled his findings into a nontechnical summary of selected literature which showed that hooking mortality of trout is variable, depending on hooking location, hooking duration, fish size, and water temperature. Hook type (barbed vs. barbless) was never addressed or acknowledged as a significant factor in his summary. Some studies and fisheries experts have questioned the efficacy of using barbless hook regulations as a management tool (Bachman 1989; Taylor and White 1992; Schill and Scarpella 1997; Schaffer and Hoffman 2002; DuBois and Dubielzig 2004; Meka 2004; Cooke and Schramm 2007). Regulating fisheries through the use of fishing gear (barbless hooks) may be ineffective in reducing injury and mortality (Cooke and Schramm 2007).

There has been substantial interest and focused research on hooking mortality based on gear, hook type, and fish species; however, there has been less effort put into evaluating the associated capture efficiencies. Although there have been studies that have compared capture efficiencies of barbed and barbless hooks (Knutson 1987; Barnhart 1990; Shaffer 2002; Dubois and Dubielzig 2004; Meka 2004; Ostrand and Siepker 2006), only Barnhart (1990) and Meka (2004) assessed the capture efficiency of artificial flies. The terminal tackle, technique, species, and methodology used in these two studies were not directly applicable to many inland coldwater fisheries managed with barbless regulations. In addition, all of these studies were conducted in a manner that allowed anglers to know what hook treatment they were using during the study. Anglers that are aware of a certain type of hook treatment may not fish with the same level of intensity (Schaffer 2002). This bias could affect angler ability to fight and land fish with an equal amount of effort while using different hook treatments.

It is critical that fisheries managers evaluate, adopt, and monitor special regulations with specific strategies and objectives. This approach will allow for assessment of the regulations and associated responses within the fishery. Adopting special regulations without specific justification, realistic goals, and measurable objectives can lead to conflicts and poor results. Many perceive special regulations as a panacea for all existing fisheries problems (AFS 2009). Unfortunately, improper use of an otherwise effective tool can result in negative angler perceptions, continued decline of fishing quality, loss of agency and professional credibility, and unrealistic angler expectations (Behnke 1987). Understanding the capture efficiency of artificial flies, both barbed and barbless, will assist fisheries managers in making informed decisions when assessing or establishing special regulations.

The rationale for choosing only artificial flies for our study was based on information from angler surveys. California Department Fish and Game (CDFG) angler survey data collected from 1999 to 2003 was analyzed to evaluate gear preference (fly, lure, bait, or a combination) by anglers fishing both streams and lakes. None of the waters chosen had an artificial-fly-only regulation. When provided the opportunity to use either bait, lures, or flies, anglers strongly preferred using flies only, in both streams (79%) and lakes (78%) compared to anglers that only used lures (17% for both streams and lakes). A small percentage (5%) of the remaining anglers used both lures and flies, bait, or their choice was unknown. These assessments were not initiated to establish statewide angler preference, but rather, to provide a rationale for using only artificial flies for the study. The vast majority of special regulation waters within California has a barbless lure or flies regulation; however, given the strong data supporting angler preferences for using artificial flies in these waters, the focal point of this study was limited to artificial flies.

We tested the capture efficiency (CE) of three artificial fly types fished with barbed and barbless hooks. Additionally, we also evaluated CE based on angler experience. Associated data on injury rates and release times were also compared to the hook treatments.

Methods

This study was conducted on both public and private waters throughout California from 2005 to 2009. To increase the probability that sufficient data were acquired within the sampling periods, high catch per unit effort waters were selected *a priori*. These waters were also chosen to diversify the locations and trout species represented in the study. Each sampling period was 4 h in duration, randomly stratified into eight, 30-m sessions. During four sessions anglers used a barbed fly and during the other four sessions anglers used a barbless fly. Hook treatment for each session was randomly stratified throughout the sampling period to reduce the possible effects for variance in angling success. This approach also made it difficult for anglers to decipher a pattern relating to the treatment they were using during the sampling period.

For the purpose of this study, CE is defined as the proportion (%) of fish landed to the number of encounters per sampling period. The initiation of an encounter was defined as the time at which an angler confirmed (1) the trout had volitionally taken the fly, (2) he or she had set the hook, and (3) there was resistance on the rod from the fish for a period no shorter than 2 s. These criteria were established to reduce false hooking and missed strike data in the study. Only sampling periods that had at least two encounters in each hook treatment were used in the analysis. Encounters in which the trout broke the line and was not landed were excluded from the analysis.

We used volunteers and CDFG personnel to conduct the angling portion of the study. Only CDFG personnel were used to observe and assist the anglers during the study. These “observers” were responsible for making the study “blind” by tying-on and switching flies for the angler. This eliminated the ability of the angler to see whether he or she was using a barbed or barbless fly. This element of the study was critical in eliminating the potential bias anglers may have had regarding the hook treatment, thus providing assumed equal effort among the sessions and encounters. At the beginning of each sampling period, the observer would randomly select one of 64 scenario cards which provided the pattern of 30-min sessions during the sampling period. These scenario cards represented all of the different potential combinations of session patterns. During the course of each sampling period, the observer would change hook treatment based on the preselected session rotation on the card. Anglers were trained, prior to sampling, in study protocols and were given the necessary field gear. Flies were separated by hook treatment and fly type into labeled fly boxes. Observers maintained possession of the fly boxes at all times and were the only ones to handle, switch, and tie-on flies during and between treatment sessions. Anglers were given the choice of what fly type they wanted to begin with. Anglers were allowed to switch fly types until the first encounter occurred. Once the first encounter occurred, the angler had to stay with that fly type for the duration of the sampling period. To allow flexibility, anglers could switch fly type patterns and or colors within that fly type during the sampling period. Hook style was limited to straight shank J-style 1–3 X long, 1 -2 X wide, sizes 16 to 6 hooks for all fly types. All flies that were obtained for the study were initially barbed; thus, flies used in the barbless hook treatment were de-barbed prior to the sampling periods. De-barbing was conducted by CDFG personnel using specially designed de-barbing pliers.

Anglers were classified as either advanced (> 200 d experience), intermediate (30 to 200 d experience), or novice (< 30 d experience) based on the total number of days they had fly fished in their life. In addition to switching and tying-on the flies for the anglers, observers also kept track of the session rotation time, type of hook treatment, duration of fight, species, hooking location on the fish, injury, and handling time. The various timed events were recorded in seconds using handheld stopwatches. Timing of the encounters by the observers began when he or she confirmed the initiation of the encounter (per the criteria stated previously) and continued until the angler landed the trout. Anglers landed all fish by use of a soft mesh net. After the angler landed the fish, they passed the fish in the net to the observer for hook removal, injury assessment, species identification, measurement in total length (TL), and notation of hooking location. For the purpose of this study, injury was defined as torn tissue, bleeding, or external hooking. Only injuries resulting from the encounter or the hook removal process were noted. Severity of injury was not assessed or ranked during the study.

We used a paired t-test to determine the difference between the CE for barbed and barbless flies for all anglers and fly types combined. A paired t-test was also used to evaluate the difference in injuries and hook treatment. The influences of angler experience and fly type to capture efficiency were evaluated independently using a one-way analysis of variance (ANOVA) general linear model procedure. A Pearson rank correlation analysis was used to evaluate the relationship of capture efficiency and number of encounters per sampling period. Statistical analysis was performed using SAS version 9.1.3 (SAS Institute 2006). All values reported are means ± one standard error (SE). All tests were assessed for significance at $\alpha = 0.05$.

RESULTS

A total of 32 different anglers were used in the study with each angler participating in one to seven different sampling periods. Although some anglers participated in multiple sampling periods, each period was treated as unique in the analysis. Eighteen different CDFG personnel served as observers. Seventy eight sampling periods out of 98 total were used in the analysis. Twenty sampling periods were removed from the analysis due to low (less than two) number of encounters per treatment. Within the 78 sampling periods, 12 encounters were removed from the analysis due to the line breaking during the encounter. Thus, a total of 2,258 encounters qualified for analysis with 48% (n = 1,077) in the barbed treatment and 52% (n = 1,181) in the barbless treatment (Table 1).

Table 1. Summary of trout landed and not landed by fly type and angler experience.

Angler experience and fly type	Barbed Flies: Trout Landed	Barbed Flies: Trout not landed	Barbed Flies: Total number of encounters		Barbless Flies: Trout Landed	Barbless Flies: Trout not landed	Barbless Flies: Total number of encounters
Advance angler							
Dry fly	308	69	377		297	114	411
Nymph	91	40	131		73	38	111
Streamer	72	18	90		72	34	106
Intermediate angler							
Dry fly	200	62	262		219	116	335
Nymph	52	28	80		42	31	73
Streamer	21	7	28		20	16	36
Novice angler							
Dry fly	66	23	89		56	27	83
Nymph	2	2	4		1	2	3
Streamer	14	2	16		11	12	23
Totals	826	251	1077		791	390	1181

A total of 1,617 trout were landed (mean TL = 213 mm ± 2.54 mm) with a range of 64 mm to 660 mm. Trout species caught during the study consisted of coastal rainbow trout *Oncorhynchus mykiss irideus*, Lahontan cutthroat trout *Oncorhynchus clarkii henshawi*, California golden trout *Oncorhynchus mykiss aguabonita*, brown trout *Salmo trutta*, and brook trout *Salvelinus fontinalis*. Hatchery raised trout made up 4% (n = 60) of the trout captured during the study, with the remaining 96% (n = 1,557) being wild trout (Table 2).

Table 2. Water locations and associated trout species landed.

Water	County	Rainbow trout	Brown trout	Brook trout	California golden trout	Lahontan cutthroat trout
Antelope Creek[1]	Shasta	21	9	2		
Antelope Creek Lake[1]	Shasta	39	5			
Bassi Fork Creek	El Dorado	78	10			
Capels Creek	El Dorado	57	5			
East Fork Carson River	Alpine	61				
Golden Trout Creek	Kern				159	
Junction Reservoir	Mono	244				
South Fork Kern River	Kern		9		3	
Mulkey Creek	Kern				104	
Owens River	Inyo	11	15			
Paiute Creek	Fresno			6	116	
Parker Creek	Mono		10	4		
Pauley Creek	Sierra	20				
Rubicon River	El Dorado	121	5			
Rush Creek	Mono	10	23			
Siberian Creek	Fresno				82	
Silver King Creek	Alpine	281				
Slinkard Creek	Mono					28
Sly Park Creek	El Dorado	6				
Middle Fork Stanislaus River	Tuolumne	51	1			
East Walker River	Mono	1	20			
	Total	1001	112	12	464	28

[1] *Denotes waters where hatchery raised rainbow trout were captured, all other trout were wild.*

The make-up of angler experience that qualified with their respective sample periods consisted of 34 advanced anglers (44%), 32 intermediate anglers (41%), and 12 novice anglers (15%). Sampling periods conducted in lotic habitats made-up the majority of the study (83%) with surveys in lentic habitats making-up a smaller portion (17%). Anglers chose to use dry flies the majority of the sampling periods (57%), followed by nymphs (33%), and streamers (10%). The mean number of encounters per sampling period was 28.9 ± 3.5, with a range of 4 to 192. Advanced anglers had the highest mean number of encounters with 36 ± 5.3, followed by intermediate anglers with 25 ± 6.1, and novice anglers with 18 ± 3.5. There was no correlation with the number of encounters per sampling period and CE (Pearson correlation coefficient: 0.17, p = 0.13). Anglers using barbed flies landed, on average, significantly (t = 4.50; df = 77; P < .0001) more trout compared to when they used barbless flies for all qualifying sampling periods. Mean CE for all angler experience level and fly types combined that used barbed flies was 0.757 ± 0.02 and 0.626 ± 0.03 for barbless flies. Capture efficiency was not significantly influenced by angler experience (ANOVA: F = 2.25; df = 2, 75; P < 0.11); however, mean capture efficiencies show a generalized decreasing trend for angler experience and barbless flies (Figure 1). Lack of significance may be due, in part, to unbalanced sample size and relative low numbers of novice anglers in the study.

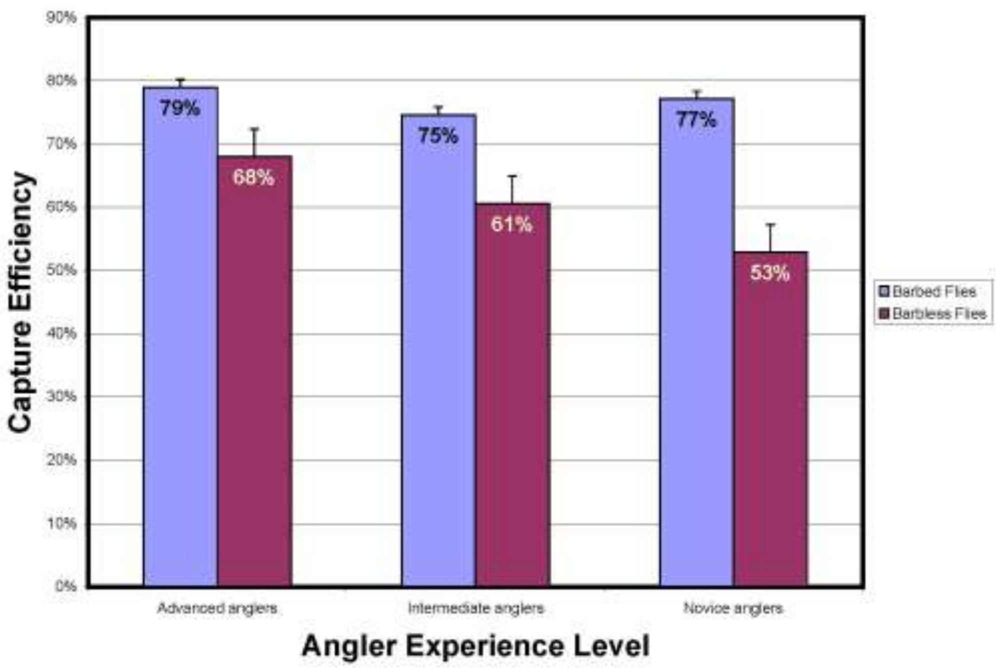


Figure 1. Mean capture efficiencies for barbed and barbless flies in association with angler experience shown as percentages with 1 SE.

Fly type had a weakly significant influence on CE (ANOVA: F = 3.24; df = 2, 77; P = 0.04); however, given the low comparative sample size for the streamer fly type, caution should be used in interpreting the results. Capture efficiency, on average, for dry flies was the highest with 0.721 ± 0.020, followed by streamers at 0.702 ± 0.054, and nymphs with 0.632 ± 0.029.

Handling times for trout caught with barbless flies, on average, was significantly lower (t = 1.66; df = 75; P < 0.0001) than trout caught with barbed flies. Mean handling times for trout caught on barbed flies was 35.5 ± 1.8 s compared to trout caught on barbless flies which was 28.3 ± 1.5 s. Injuries sustained from barbed flies were, on average, significantly higher than barbless flies (t = 4.71; df = 75; P < 0.0001). The proportion of injuries, on average, for trout landed and handled with barbed flies was 0.24 compared to barbless flies which were 0.12. . No direct mortality was observed for any of the trout landed during the study. Two sampling periods were excluded from the analysis for handling time and injuries. This was done because ratios for injury and handling times could not be established due to absence of landed trout in one of the treatments. It should be noted that the absence of landed trout did not exclude the two sampling periods from any of the CE analysis.

Discussion

Results from this study show that, on average, barbed flies effectively land more trout than barbless flies, regardless of fly type and angler experience. Although this may seem intuitive for both fisheries managers and anglers alike, the significance of the difference highlights the need for fisheries managers to justify the efficacy of barbless regulations based on biological rationale and management objectives. Some angler groups may gauge their satisfaction with a given angling experience by how many fish they catch. This may be in the form of fish that are caught and released or ones that are harvested. Reducing catch rates through the use of barbless regulations will likely affect all anglers, regardless of experience, ability to land and harvest trout if they are using artificial flies.

There are over 80 waters throughout California, not including one entire county (San Diego), that have barbless regulations which allow for harvest of one or more trout. These regulations reflect both anadromous and inland trout waters and range from seasonal to year-round restrictions. Based on our results, anglers who volitionally want to harvest trout in these waters will face a significantly reduced capture efficiency using barbless flies. These regulations could result in reduced angler catch rates and associated decreased angler satisfaction. Anglers interested in harvesting trout in these barbless regulated waters will also likely have to spend more time angling in an effort to harvest trout, thus increasing the number of anglers on the water at any given time. Reduced catch rates may have substantial direct and indirect effects on angler satisfaction in waters that already have high angler use and low capture rates.

Reduced catch rates from barbless hooks may also affect directed management objectives. In some cases, fisheries managers may want harvest on some portions of a fishery. This is the case for a number of anadromous steelhead fisheries in California that are supplemented with hatchery fish. All of the hatchery steelhead stocked in California are marked with an adipose clip to allow differentiation from wild steelhead. There is currently no allowable harvest on wild steelhead in California; however, harvest of hatchery steelhead is promoted by fisheries managers. Recent regulatory changes have increased the bag limits for hatchery steelhead throughout many California coastal waters based on this approach. Catch rates in steelhead waters can be very low and any reduction in catch rates, especially for hatchery fish, will likely have an effect on management objectives (reduced average harvest on hatchery fish) and angler satisfaction (less fish landed on average).

Reduced catch rates from barbless hook regulations may not have the same effect on all angling groups in relation to their overall angling experience. Novice anglers showed both the lowest mean encounter rate and the lowest mean CE (53%) of all the angler experience levels in the study. Providing the best possible angling experience for novice anglers may prove essential in maintaining their interest in the sport. In evaluating affects of catch rates and associated satisfaction within specific user groups, it may prove very important for young anglers (< 15 years) and consumptive anglers (interested in harvest) to get their bag limit during a fishing outing (Sanyal and McLaughlin 1993). Capture efficiency for advanced anglers was the highest overall, along with showing the smallest average difference (11%) in CE between the two hook treatments. Advanced anglers may not have as much of a concern about barbless regulations given the small, yet significant, difference in CE for them as an angling group. This proposed lack of concern could be especially true for advanced anglers who are not interested in harvest. The challenge facing fisheries managers is to assess what the objective and goals are regarding managing a fishery, for all different types of angling groups.

Although the results from our study show a statistically significant difference in mean handling times between barbed and barbless flies, the difference is likely not biologically significant. The difference in mean handling times between the different hook treatments was only 7.2 s, which would not likely lead to increased mortality. This assessment is similar to DuBois and Dubielzig (2004) who found that the slight decrease in unhooking times gained by using barbless single-hook spinners is unlikely to improve the survival chances or reduce sublethal injuries of released fish. All of the trout captured in our study were landed and processed with the use of a soft mesh landing net. This probably reduced both encounter times and handling times. In comparison to Meka (2004), our mean handling times were nearly half of what they reported. These reduced times may be due to the relatively mean small size (213 mm ± 2.54) of the trout captured, and because handling fish was conducted by experienced CDFG personnel, not the anglers. It makes intuitive sense that by using a landing net, anglers could substantially limit the stress from the encounter by reducing landing and the handling time. Injury rates were relatively high for both hook treatments; however, the criteria for qualifying injury was very conservative. Although injuries were not ranked by severity, no injuries were noted in vital organs or in areas that would likely lead to post-release mortality.

ACKNOWLEDGEMENTS

I want to thank the many volunteer anglers who fished long and hard and the CDFG Heritage & Wild Trout Statewide Crew for all their support both in angling and serving as the observers. I also want to thank Ed Pert for his support in initiating and help designing the study. Jeff Weaver and Stephanie Mehalick provided critical assistance in planning and implementing the fieldwork. I also want to thank Donn Burton for help in data analysis and data entry.

REFERENCES

AFS (American Fisheries Society). 2009. Position Statement # 28: Special fishing regulations for managing fresh water sport fisheries. American Fisheries Society. Available: www.fisheries.org/afs/docs/policy_28f.pdf (January 2009).

Bachman, R. A., H. J. Stinefelt, and C. R. Gougeon. 1989. Wild trout management in the eastern megalopolis. Pages 141–152 *in* F. Richardson and R. H. Hamre, editors. Wild trout IV. Trout Unlimited, Arlington, Virginia.

Barnhart, R. A. 1990. Comparison of steelhead caught and lost by anglers using flies with barbed or barbless hooks in the Klamath River, California. California Fish and Game 76(1):43–45.

Behnke, R. J. 1987. Catch and release: the last word. Pages 291 – 298 in R. A. Barnhart and T. D. Roelofs, eds. Proceedings of the symposium on the catch and release fishing: a decade of experience. Arcata, Calilfornia.

Cooke, S. J., and H. L. Schramm. 2007. Catch-and-release science and its application to conservation and management of recreational fisheries. Fisheries Management and Ecology 14: 73-79.

DuBois, R. B., and R. R. Dubielzig. 2004. Effect of hook type on mortality, trauma, and capture efficiency of wild stream trout caught by angling with spinners. North American Journal Fisheries Management 24:609-616.

Falk, M. R., D. V. Gillman, and L. W. Dahlke. 1974. Comparison of mortality between barbed and barbless hooked lake trout. Canada Department of the Environment, Fisheries and Marine Service, Resource Management Branch Technical Report CEN/T-74–1.

Faragher, A. R. 2004. Hooking mortality of trout: a summary of scientific studies. Fisheries Research Report Series: 9. NSW Fisheries. Australian Department of the Environment and Water Resources.

Hunsaker, D., II, L. F. Marnell, and F. P. Sharpe. 1970. Hooking mortality of cutthroat trout at Yellowstone Lake. Progressive Fish-Culturist 32:231–235.

Knutson, A. C., Jr. 1987. Comparative catches of ocean sport-caught salmon using barbed and barbless hooks and estimated 1984 San Francisco Bay area charter boat shaker catch. California Fish and Game 73:106–116.

Meka, J. M. 2004. The Influence of hook type, angler experience, and fish size on injury rates and the duration of capture in an Alaskan catch-and-release rainbow trout fishery. North American Journal of Fisheries Management 24:4, 1309-1321

Mongillo, P. E. 1984. A summary of salmonid hooking mortality. Washington Department of Game, Fisheries Management Division, Olympia.

Ostrand, K. G., M. J. Siepker. 2006. Capture efficiencies of two hook types and associated injury and mortality of juvenile muskellunge angled with live bait. North American Journal of Fisheries Management 26: 622–627.

Sanyal, N., and W. J. McLaughlin. 1993. Angler market segmentation, angler satisfaction, and activity persistence among Idahoans. Idaho Fish and Game, 1992 Annual SFRA Report Project F -73-R-14, Subproject VI, Study IV. Submitted by Department of Resource Recreation and Tourism, College of Forestry, Wildlife, and Range Sciences, University of Idaho.

SAS Institute Inc. 2006. Base SAS® 9.1.3 Procedures Guide, Second Edition, Volumes 1, 2, 3, and 4. Cary, NC: SAS Institute Inc.

Schaeffer, J. S., and E. M. Hoffman. 2002. Performance of barbed and barbless hooks in a marine recreational fishery. North American Journal of Fisheries Management 22:229–235.

Schill, D. J., and R. L. Scarpella. 1997. Barbed hook restrictions in catch-and-release trout fisheries: a social issue. North American Journal of Fisheries Management 17:873–881.

Taylor, M. J., and K. R. White. 1992. A meta-analysis of hooking mortality of nonanadromous trout. North American Journal of Fisheries Management 12: 760–767.

Wydoski, R. S. 1977. Relation of hooking mortality and sublethal hooking stress to quality fishery management. Pages 43–87 *in* R. A. Barnhart and T. D. Roelofs, editors. Catch-and-release fishing as a management tool. Humboldt State University, California, Cooperative Fishery Research Unit, Arcata.

MIGRATORY DISTRIBUTION OF FLUVIAL ADULT BULL TROUT IN RELATION TO LAND AND WATER USE IN WATERSHEDS WITHIN THE MID-COLUMBIA AND SNAKE RIVER BASINS

S. J. Starcevich[1], P.J. Howell[2], S. E. Jacobs[1], and P. M Sankovich[3]

[1]Oregon Department of Fish and Wildlife, ODFW Research Lab, 28655 Highway 34, Corvallis, Oregon, 97333, USA

[2]USDA Forest Service, Forestry and Range Sciences Laboratory, 1401Gekeler Lane, La Grande, Oregon, 97850, USA

[3]U.S. Fish and Wildlife Service, Columbia River Fisheries Program Office, La Grande Field Office, 3502 Highway 30, La Grande, Oregon, 97850, USA

ABSTRACT — Radio telemetry was used to investigate migratory patterns of 198 fluvial adult bull trout *Salvelinus confluentus* (mean, 449 mm FL) in relation to land and water use characteristics in the Imnaha, Grande Ronde, Walla Walla, Umatilla, and John Day river basins. Median migration distances of radio-tagged fluvial bull trout from the Imnaha (89 km) and Wenaha (56 km) and Lostine (41 km) rivers were relatively long. These study areas were characterized by low levels of water consumption, private landownership, and population density. Median migration distances were significantly shorter in the John Day (8 km) and Umatilla (22 km) rivers and Mill Creek (20 km), which were characterized by greater water and land use and no known barriers to movement. Bull trout from the Lostine and Wenaha rivers also returned to habitats in winter that were more extensive (73 and 86 km, respectively), and the spawning and wintering areas were spatially separated. In contrast, winter locations of bull trout in the John Day, Walla Walla, and Umatilla river were distributed over a relatively short mainstem reach (<25 km) adjacent to or overlapping the spawning distribution. These results suggest adult fluvial migration may be restricted in basins with substantial water and land use. Additional research on bull trout ecology in larger rivers and the effects of anthropogenic habitat degradation on the spatiotemporal distribution of critical resources will lead to a better understanding of how migrations patterns are established and the factors that limit distribution.

INTRODUCTION

Bull trout *Salvelinus confluentus*, and chars in general, are considered glacial relicts and have evolved several life history traits advantageous for persistence during glacial expansion and for recolonization of suitable habitat during glacial retreat (Power 2002). Among these traits are the physiological adaptation to cold water and the ability to move within freshwater systems to find resources (Northcote 1997; Power 2002). Adults spawn and juveniles rear within the coldest sections of the stream network, which are usually small, high-elevation headwater streams (Rieman and McIntyre 1993). During the fluvial subadult stage, when juveniles disperse from their early rearing habitat and their first spawning migration (Muhfeld and Marotz 2005), bull trout may spend 1 to 3 years rearing to adulthood in larger river habitats (Mogen and Kaeding 2005). These habitats provide greater space and food resources, which improve growth and reproductive potential (Gross 1987; Northcote 1997). Fluvial (e.g., Bjornn and Mallet 1964) and adfluvial (e.g., Fraley and Shepard 1989) bull trout can migrate over 250 km between their spawning grounds and these productive river and lake habitats.

Diversity in migratory behavior is important to the stability and persistence of bull trout populations (Rieman and McIntyre 1993). The extent and variation of fluvial migrations reflect how local populations have adapted to the spatiotemporal distribution of local habitats (Southwood 1977) and may provide information on how life history expression is affected by anthropogenic habitat alteration and degradation (Rieman and McIntyre 1995; Dunham and Rieman 1999). We used radio telemetry to study migration distance and diversity of fluvial adult bull trout in a range of habitat conditions in northeastern Oregon and southeastern Washington, a region for which there was little information on bull trout migrations. The study region included study areas with extensive human land and water use and those with substantially less human influence. Specifically, our objective was to quantify migration and seasonal distribution patterns of fluvial adult bull trout in relation to the wide range of human land and water use in our study areas.

Study Area

Bull trout were radio tagged in the Imnaha, Wenaha, Lostine, John Day, and Umatilla rivers and in Mill Creek (Figure 1). This region generally has a semiarid, continental climate and most precipitation falls as snow at higher elevations from November to May. In the Umatilla and Walla Walla river basins, the climate is modified by marine air from the Pacific Ocean, which brings rain in late fall and winter. The known bull trout spawning distribution was located in the forested headwaters of these watersheds in areas that are generally under federal management, most of which was in wilderness areas or other protective designations with little development (e.g., municipal watersheds) (USFWS 2002).

The floodplain habitat below the spawning reaches in the John Day (basin area, 20,980 km^2) and Umatilla (6,580 km^2) rivers and Mill Creek (Walla Walla basin area, 4,450 km^2) has been altered by over a century of human activities (USFWS 2002) that have resulted in the extensive loss of riparian vegetation, channel complexity, in-stream large wood, and large pools (Wissmar et al. 1994). In the Umatilla and Walla Walla river basins, seasonal dewatering of large river sections was common historically and has been reported as recently as 2000 (USFWS 2002). In Mill Creek, Bennington Diversion Dam (river kilometer [RK] 18) was originally built in 1942 with no fish passage facilities and was retrofitted with a fish ladder in 1982. Through the city of Walla Walla, Mill Creek

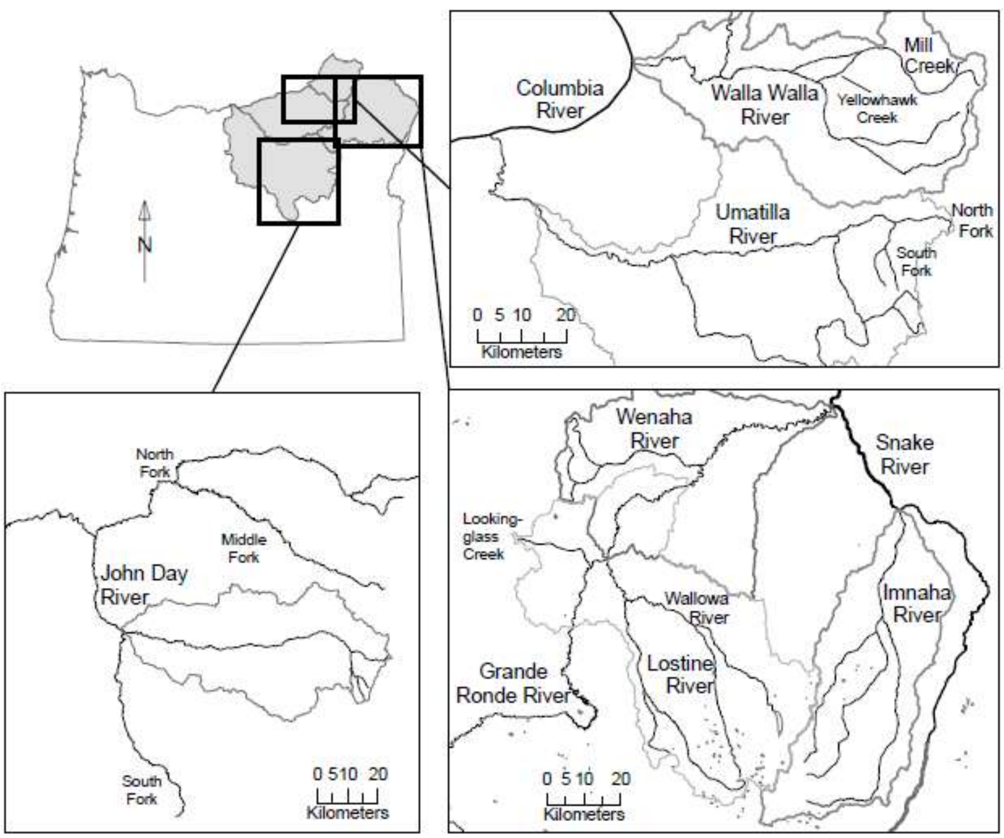


Figure 1. The study region in the Columbia and Snake river basins and individual study areas (outlined in gray).

is a concrete canal with channel-spanning weirs.During summer most of the river is diverted into Yellowhawk Creek, which is a modified irrigation diversion that provides an additional connection to the Walla Walla River. Unscreened diversions on Yellowhawk Creek may obstruct upstream fish passage and entrain fish moving downstream (USFWS 2002)

Similarly, migratory reaches downstream of spawning habitat in the Lostine River (basin area, 240 km^2)and portions of the Wallowa River (RK 20 upriver to Wallowa Lake) have reduced habitat quality because of irrigation diversions, other agricultural practices, residential development, and stream channelization (Howell et al. 2010; USFWS 2002). In contrast, the lower reaches of the Imnaha, Grande Ronde, and Snake rivers potentially used by migratory bull trout have relatively low human population density and high summer base flows (respective means, 21 and 504 cms).

Methods

Fish Capture

Bull trout were caught in the Imnaha River by angling between RK 98 and 107. Fish were captured in the Wenaha River by angling near RK 5, 14, and 20. In the Lostine River, bull trout were caught in an upstream picket weir trap near the mouth (RK 1) and by angling between RK 17 and 39. In the John Day River basin, fish were caught by angling and in weir traps in Call Creek (RK 0.5), Deardorff Creek (RK 5), Roberts Creek (RK 1), and two location on the mainstem (RK 437 and 450). Bull trout were captured by angling between RK 140 on the upper Umatilla River and RK 2 on the North Fork Umatilla River. In Mill Creek, bull trout were caught by angling in the pools adjacent to the municipal intake dam (RK 41) or in a trap affixed to the upstream end of its fish ladder.

Radio Transmitters and Tagging

We used radio transmitters that ranged in battery life from 8 to 24 months (Lotek NTC-6-2, and Advanced Telemetry Systems models 2-357, 2-375, and 10-28) and emitted a pulsed signal at frequencies from 150 to 152 MHz. Transmitter weight did not exceed 3% of the host fish weight. Bull trout were anesthetized prior to and during surgery with 50 mg/L tricaine methanesulfonate buffered with equal amounts of sodium bicarbonate. The transmitters were implanted into the peritoneal cavity using the methods described by Winter (1996). The transmitter antenna was passed through the body wall using a technique similar to that described by Ross and Kleiner (1982). Surgery lasted less than six minutes. The fish recovered from anesthesia in a covered and aerated bath for at least 15 min before being released in slow, deep water near the capture site. Fish were not tagged when water temperatures exceeded 15°C.

Radio Tracking

Radio-tagged fish were tracked from the ground and air using a Lotek receiver (SRX 400). We used a handheld two-element antenna when tracking on foot and a five-element Yagi antenna when tracking by vehicle. Aerial tracking was conducted from a Cessna 180 with two-element antennas affixed to each wing. When tracking by vehicle or foot, we estimated the transmitter location in the river by triangulating on the strongest signal (White and Garrot 1990). We estimated aerial tracking error by comparing aerial location estimates with the actual location of transmitters we placed in the river. The interval between tracking observations differed among watersheds depending on remoteness, private land accessibility, and flight availability.

Quantification of Migration and Distribution

We quantified migration distance as the distance in river length between the farthest upstream location during the spawning period and winter modal location. Winter modal location represented where a fish was observed most often during winter. We plotted on maps these two locations for each fish to show seasonal distribution patterns. The spawning period, based on previous spawning surveys in these basins, was defined as 15 August to 15 November.

Water and land use index

We used median water consumption in summer as an index to evaluate the relationship between human land and water use and median migration distance, because it represented water consumption as well as irrigated agricultural acreage and the degree of urbanization in the large river habitats in each study area. It was indeed significantly positively correlated with two measures of general human land use: private landownership (Pearson product correlation coefficient [r] = 0.89; P=0.017) and, after square root transformation, population density (r = 0.84; P=0.036).

Water use was characterized in our study areas using the online Water Availability Reporting System [WARS] provided by the Oregon Water Resources Department website. We used the estimated reduction in the median monthly natural streamflow (i.e., a flow that is exceeded 50% of the time for a particular month) caused by consumption of surface water at several WARS stations during July through September (see Cooper [2002] for a detailed description of this calculation). We selected the WARS station nearest the downstream end of each study area. These stations were located at the mouth of the Imnaha, Grande Ronde, and Wallowa rivers; and at RK 342 of the John Day River, RK 66 of the Umatilla River, and RK 77 of the Walla Walla River.

Private landownership percentage and population density were estimated using a geographical information system [GIS] and coverages for private and public (i.e., federal and state) landownership and population density (2000 census) in 5th field hydrologic units adjacent to the potential migratory habitat of each population. This was defined as 100 river km from the lower limit of the observed spawning distribution. We selected the 100 km measurement for all study areas because it represented the longest migrations observed in this study and facilitated stndard comparisons among basins.

Data Analysis

To determine if there were significant differences (P<0.05) among the basins in migration distance, we used the Kruskal-Wallis test on ranks (Sokal and Rohlf 1995) as the Kolmogorov-Smirnov test (with Lilliefor's correction) indicated the data were not normally distributed. We compared individual basins using Dunn's method (Dunn 1964) for multiple comparisons of ranked data and unequal sample sizes. Pearson product-moment correlation (Sokal and Rohlf 1995) was used to evaluate the relationship between the water and land use index and median migration range.

Results

Radio Tagging and Tracking

We radio tagged 198 adult bull trout in 6 basins (Table 1). Fish fork length (FL) averaged 449 mm and ranged from 260 to 675 mm. Ninety-three percent of the fish were tagged between March and early September and 7% were tagged in October and November. Overall, 51% were tracked through spawning and at least one winter (Table 1). The time between observations ranged from 7 d in the Imnaha River and Mill Creek; 11 to 22 d in the Lostine, John Day, and Umatilla rivers; to 25 d in the Wenaha River. The longer interval in the Wenaha River was caused by the relative inaccessibility of the watershed and the difficulty in obtaining tracking flights. The mean tracking error from comparing aerial location estimates (N=15) to known transmitter locations in Mill Creek was 1.7 km (range, 0.2 to 3.1 km). The error associated with tracking by vehicle or on foot was not determined but presumably was much less than aerial tracking error.

Table 1. Study period, number of bull trout tagged, fork length (FL) mean and range, and the number (and percentages) of fish tracked through at least spawning and the first winter.

Study area	Year	N	Mean FL (mm)	Range FL (mm)	≥1st winter *N* (%)
John Day River	1998-1999	23	405	285-560	17 (74)
Lostine River	2001, 2004	41	468	360-600	14 (34)
Wenaha River	1997-1999	51	461	260-645	40 (78)
Mill Creek	1997-1999	46	441	282-630	20 (43)
Umatilla River	2002	15	410	351-513	7 (47)
Imnaha River	2001	22	470	379-675	3 (15)
Totals		198			101 (51)

Migration Distance in Relation to Water and Land Use

We found a significant negative correlation (r = -0.89; P=0.019) between median migration distance and the water and land use index. The median migration distances were relatively high in the Imnaha (median, 89; range, 89-116), Wenaha (median, 56 km; range, 11-100), and Lostine (median 41 km; range, 6-77) rivers. These basins had relatively low levels of water and land use (Table 2). Shorter median migration distances were observed in the John Day River basin (median, 8 km; range, 1-46), Mill Creek (median, 20 km; range, 6-31) and Umatilla River basin (median, 22 km; range, 9-33). These basins showed greater levels of water and land use.

Table 2. Characteristics of human influence in each study area.

Study area	Median water consumption in summer (%)	Private land (%)	Population density (Pop./km2)
Imnaha River	17	29	0.1
Wenaha River	27	44	0.1
Lostine River	34	53	1.1
John Day River	81	55	1.7
Umatilla River	84	72	5.8
Mill Creek	95	86	20.0

There were significant differences (df = 5, P<0.001) in median migration distance among fish from different study areas (Figure 2). Specifically, the median migration distances of fish from the Imnaha, Wenaha, and Lostine rivers were not significantly different from each other (P>0.05). Bull trout from the Imnaha and Wenaha river basins showed significantly longer median migration distances (P<0.05) than the other basins, while the Lostine River fish were significantly different only from those in the John Day River basin. There were no significant differences (P>0.05) in median migration range among bull trout from the John Day and Umatilla river basins and Mill Creek.

During the spawning period, adults were distributed in the upper watershed of each basin (Figure 3 and 4). Winter distribution of adults from the Lostine and Wenaha rivers were extensive (73 and 86

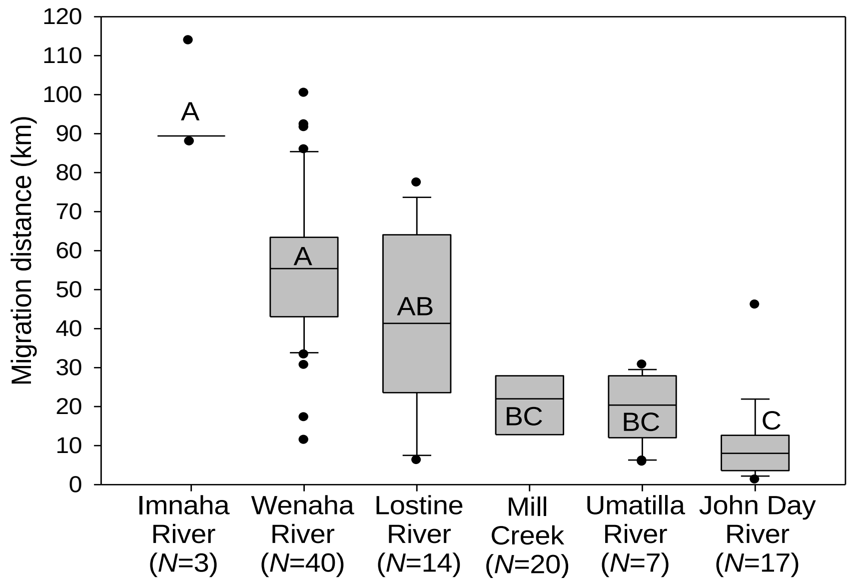


Figure 2. Box plots of annual range with median (solid line), two middle quartiles (box), 5th and 95th percentiles (whiskers), and outliers (black dots) for tagged bull trout in each study area. Letters denote significant differences (P<0.05) among study areas.*Seasonal distribution*

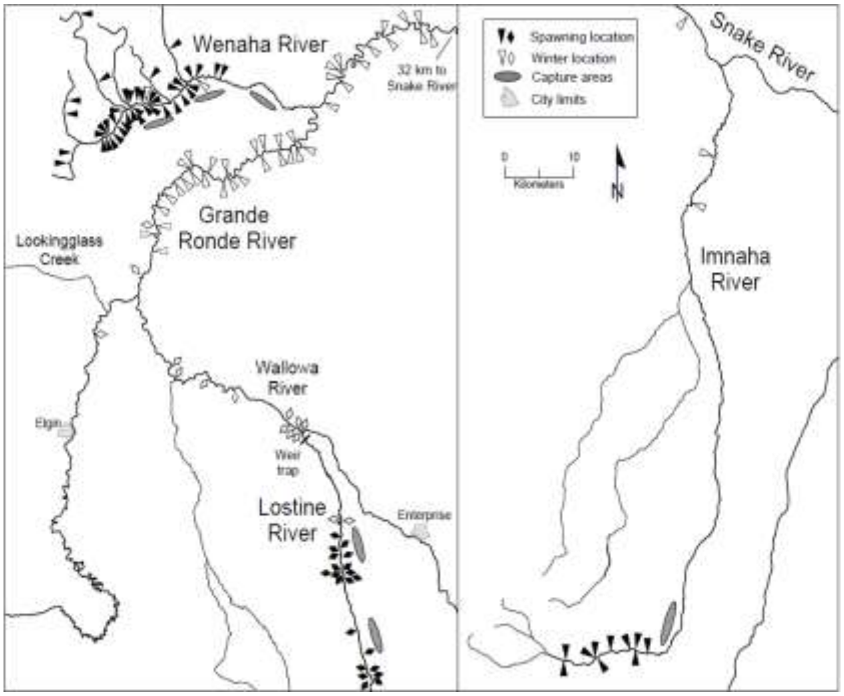


Figure 3. Migratory distribution of bull trout in the study areas of the Wenaha, Lostine, and Imnaha rivers.

km, respectively) and included larger river habitats. In contrast, winter distributions were more restricted for bull trout from Mill Creek (21 km), Umatilla River (24 km), and the John Day River (49 km). In the John Day River, 93% of bull trout were distributed in winter in the upper 13 km.

Discussion

The migratory distribution of fluvial adult bull trout among the six river basins was generally related to differences in water and land use characteristics of the basins. Bull trout in the Grande Ronde and Imnaha rivers migrated significantly greater distances between the spawning and overwintering areas (medians, 41-89 km; maxima, 77-116 km) than in the other basins (medians, 8-22 km; maxima, 31-46 km), which were more highly impacted by water and land use practices. In the interior Columbia River basin, bull trout occurrence and population strength were similarly inversely related to road density and intensity of development (Rieman et al. 1997; Lee et al. 1997).

The migratory behavior of fish from the Imnaha and Grande Ronde rivers was similar to that reported in other watersheds with no passage barriers and relatively little human influence. For example, in the Salmon River basin, Idaho, bull trout migrated between 35 and 106 km in the South Fork basin (Schill et al. 1994; Hogen and Scarnecchia 2006); and in the Middle Fork basin, Bjornn and Mallet (1964) recorded seasonal movements up to 307 km. In the complex and woody Morice River, tributary of the Skeena River, British Columbia, migrations of radio-tagged bull trout extended over 75 km (Bahr and Shrimpton 2004). In the Athabasca River, Alberta, McLeod and Clayton (1997) recorded adult annual ranges between 59 and 110 km.

In contrast, the migration distances we observed in Mill Creek and the John Day and Umatilla river basins are uniquely short when compared to the published literature on large-bodied (>300 mm FL) fluvial bull trout. In most other instances, the reduction or loss of diversity in migratory behavior has been associated with dam construction, nonnative species or habitat degradation (e.g., Fitch 1997, Swanberg 1997, Jakober et al. 1998, Brenkman et al. 2001, Nelson et al. 2002). The rarity, or lack, of long distance migration among tagged bull trout in three of our study areas suggests that fluvial life history expression has been curtailed; however, there is little information in these study areas about the spatiotemporal distribution of resources critical to bull trout life history expression and the effect of human activities on them.

The distributions of wintering locations also contrasted sharply between the study areas. Most bull trout from the Wenaha and Lostine rivers returned to larger rivers in winter and were spread over long distances (73-86 km) and generally had some spatial separation between spawning and wintering areas, similar to the patterns observed for fluvial bull trout in the upper Salmon River basin (e.g., Schill et al. 1994; Hogen and Scarnecchia 2006; Watry and Scarnecchia 2008). These two study areas also contained diverse migration patterns. Two large-bodied bull trout from each study area displayed short migrations (6-17 km) and resided year-round within the known spawning distribution and most of the

Wenaha River migrants (66%) displayed a unique fluvial migration pattern. This pattern consisted of a post-spawning migration downstream out of the Wenaha River and then upstream to wintering areas in the Grande Ronde River as far as 49 km from the Wenaha River confluence. This postspawning migration pattern differed from the downstream-only patterns previously reported for adult fluvial bull trout (e.g., Bjornn and Mallet 1964; Jakober et al. 1998; Bahr and Shrimpton 2004; Hogen and Scarnecchia 2006) and was similar to those observed in adfluvial populations (e.g., Herman 1997, Hogen and Scarnecchia 2006, DuPont et al. 2007, Watry and Scarnecchia 2008). As DuPont et al. (2007) noted for adfluvial bull trout populations, the complex pattern we observed expands our view of what may have been historically occupied habitats for fluvial bull trout populations.

In contrast, winter locations of fluvial bull trout from Mill Creek and the Umatilla River were distributed over a relatively short main stem reach (<25 km). In the John Day River basin, 93% of the fish were distributed over a 13 km main stem reach in winter. Although fish in these basins generally alternated between headwater stream habitat during the spawning period and larger main stem habitats in winter, there was little or no spatial separation between wintering and spawning areas. Small numbers of subadult bull trout have been reported downstream of the adult winter distributions we observed in other studies in the John Day River near

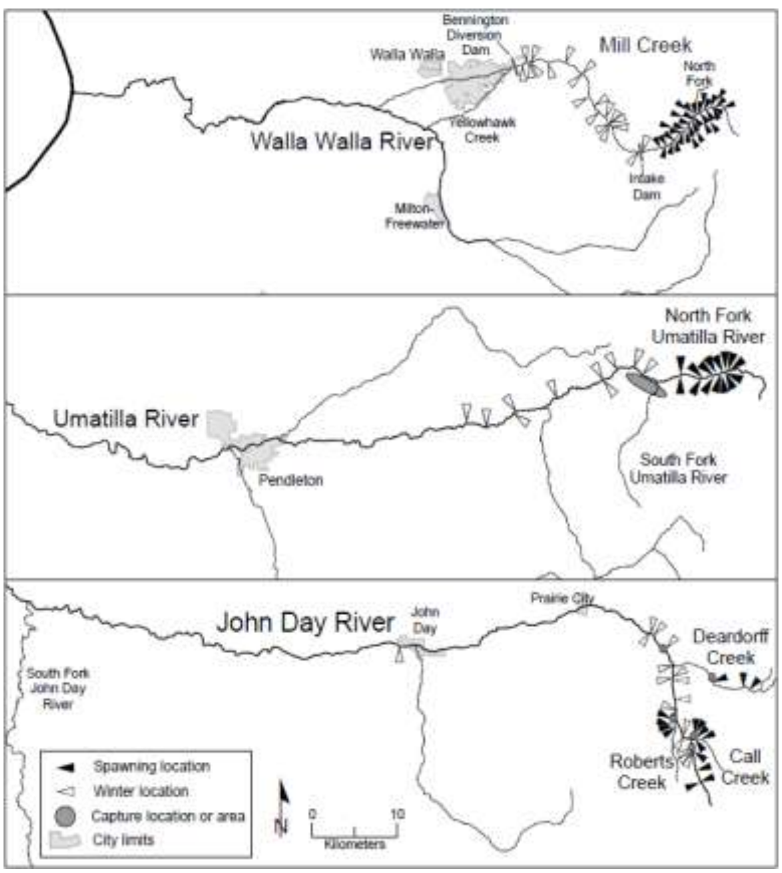


Figure 4. Migratory distribution of bull trout in the study areas of Mill Creek and the Umatilla and John Day rivers.

the South Fork John Day River confluence (Wilson et al. 2008), from Bennington Dam on Mill Creek to the lower Walla Walla River (Anglin et al. 2009), and in the lower Umatilla River (RK 5) (P. Bronson, Confederated Tribes of the Umatilla Indian Reservation, personal communication, 2008). These studies show that in these study areas the distribution of subadult bull trout may be more extensive than that of the adults in our study; however, there is no information on subadult survival in these habitats.

Additional research on juvenile and adult bull trout habitat selection in larger rivers and the effects of anthropogenic habitat degradation on the spatio-temporal distribution of critical resources will lead to a better understanding of the factors that limit migratory distribution.

Acknowledgements

We thank Dave Buchanan and Al Hemmingsen for their role in planning and conducting this research. We thank Larry Boe, Jason Shappart, Blane Bellerud, Ian Tattam, Lisa and Dave Gaudette, and Chris Abbes for field assistance; Glen Mendel (Washington Department of Fish and Wildlife) Tim Bailey and Tim Unterwegner (ODFW), and Dave Crabtree (U.S.D.A. Forest Service [USFS]) for their assistance and cooperation; and the Oregon State Police for piloting the plane during tracking flights. This research was funded by the Bonneville Power Administration and the USFS.

References

Anglin, D. R., D. Gallion, M. Barrows, R. Koch, and C. Newlon. 2009. Monitoring the Use of the Mainstem Columbia River by Bull Trout from the Walla Walla Basin. 2009 Annual Report to the U.S. Army Corps of Engineers, Walla Walla District. Project BT-W-05-6.

Bahr, M. A., and J. M. Shrimpton. 2004. Spatial and quantitative patterns of movement in large bull trout (*Salvelinus confluentus*) from a watershed in north-western British Columbia, Canada, are due to habitat selection and not differences in life history. Ecology of Freshwater Fish 13:294-304.

Bjornn, T. C., and J. Mallet. 1964. Movements of planted and wild trout in an Idaho river system. Transactions of the American Fisheries Society 93:70-76.

Brenkman, S. J., G. L. Larson, and B. E. Gresswell. 2001. Spawning migration of lacustrine-adfluvial bull trout in a natural area. Transactions of the American Fisheries Society 130:981-987.

Cooper, R. M. 2002. Determining surface water availability in Oregon. State of Oregon Water Resources Department, Open File Report SW 02-002.

Dunham, J. B., and B. E. Rieman. 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. Ecological Applications 9:642-655.

Dunn, O. J. 1964. Multiple contrasts using rank sum tests. Technometrics 6:241-252.

DuPont, J. M., R. S. Brown, and D. R. Geist. 2007. Unique allacustrine migration patterns of a bull trout population in the Pend Oreille River Drainage, Idaho. North American Journal of Fisheries Management 27:1268-1275.

Fitch, L. 1997. Bull trout in southwestern Alberta: notes on historical and current distribution. Pages 427- 434 *in* W. C. Mackay, M. K. Brewin, and M. Monita, editors. Friends of the Bull Trout conference proceedings, Bull Trout Task Force (Alberta), Trout Unlimited Canada, Calgary.

Fraley, J. J., and B. B. Shepard. 1989. Life history, ecology, and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake and river system, Montana. Northwest Science 63:133- 143.

Gross, M. R. 1987. Evolution of diadromy in fishes. Pages 14-25 *in* M. J. Dadswell and five coeditors. Common strategies of anadromous and catadromous fishes. American Fisheries Society, Symposium 1, Bethesda, Maryland.

Herman, S. J. 1997. The unique bull trout spawning population of Pinto Lake, Alberta. Pages 217-226 in W. C. Mackay, M. K. Brewin, and M. Monita, editors. Friends of the Bull Trout conference proceedings. Bull Trout Task Force (Alberta), Trout Unlimited Canada, Calgary.

Hogen, D. M., and D. L. Scarnecchia. 2006. Distinct fluvial and adfluvial migration patterns of a relict charr, *Salvelinus confluentus*, stock in a mountainous watershed, Idaho, USA. Ecology of Freshwater Fishes 15:376-387.

Jakober, M. J., T. E. McMahon, R. F. Thurow, and C. G. Clancy. 1998. Role of stream ice on fall and winter movements and habitat use by bull trout and cutthroat trout in Montana headwater streams. Transactions of the American Fisheries Society 127:223-235.

Lee, D. C., J. Sedell, B. E. Rieman, R. F. Thurow, J. E. Williams, and others. 1997. Broadscale assessment of aquatic species and habitats. In: Quigly, T. M., and S. J. Arbelbide, eds. An assessment of ecosystem components in the interior Columbia River Basin and portions of the Klamath and Great Basins. Gen. Technical Report PNW-GTR-405. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon.

McLeod, C. L., and T. B. Clayton. 1997. Use of radio telemetry to monitor movements and obtain critical habitat data for a fluvial bull trout population in the Athabasca River, Alberta. Pages 413-420 *in* W. C. Mackay, M. K. Brewin, and M. Monita, editors. Friends of the bull trout conference proceedings. Trout Unlimited Canada, Bull Trout Task Force (Alberta), Calgary.

Mogen, J. T., and L. R. Kaeding. 2005. Identification and characterization of migratory and nonmigratory bull trout populations in the St. Mary River drainage, Montana. Transactions of the American Fisheries Society 134:841-852.

Muhlfeld, C. C., and B. Marotz. 2005. Seasonal movement and habitat use by subadult bull trout in the upper Flathead River System, Montana. North American Journal of Fisheries Management 25:797-810.

Nelson, M. L., T. E. McMahon, and R. F. Thurow. 2002. Decline of the migratory form in bull charr, *Salvelinus confluentus*, and implications for conservation. Environmental Biology of Fishes 64:321- 332.

Northcote, T. G. 1997. Potamodromy in Salmonidae: living and moving in the fast lane. North American Journal of Fisheries Management 17:1029-1045.

Power, G. 2002. Charrs, glaciations, and seasonal ice. Environmental Biology of Fishes 64:17-35.

Rieman, B. E., and J. D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. U.S. Forest Service, Intermountain Research Station, General Technical Report INT-302, Ogden, Utah.

Rieman, B. E., and J. D. McIntyre. 1995. Occurrence of bull trout in naturally fragmented habitat patches of varied size. Transactions of the American Fisheries Society 124:285-296.

Ross, M. J., and C. F. Kleiner. 1982. Shielded-needle technique for surgically implanting radio-frequency transmitters in fish. Progressive Fish-Culturist 44:41-43.

Schill, D., R. Thurow, and P. Kline. 1994. Seasonal movement and spawning mortality of fluvial bull trout in Rapid River, Idaho. Boise, ID: Idaho Department of Fish and Game. Project: F-73-R-15, Job 2.

Sokal, R. R. and F. J. Rohlf. 1995. Biometry: the principles and practice of statistics in biological research. 3rd edition. W. H. Freeman and Co.: New York.

Southwood, T. R. E. 1977. Habitat, the templet for ecological strategies? Journal of Animal Ecology 30:1-8.

Swanberg, T. 1997. Movements of and habitat use by fluvial bull trout in the Blackfoot River, Montana. Transactions of the American Fisheries Society 126:735-746.

USFWS (U.S. Fish and Wildlife Service). 2002. Bull Trout (*Salvelinus confluentus*) Draft Recovery Plan, Portland, Oregon.

Watry, C. B., and D. L. Scarnecchia. 2008. Adfluvial and fluvial life history variations and migratory patterns of a relict charr, Salvelinus confluentus, stock in west-central Idaho, USA. Ecology of Freshwater Fishes 17:231-243.

White, G. C. and R. M. Garrot. 1990. Analysis of wildlife radio-tracking data. Academic Press, New York.

Wilson, W. H, T.L. Schultz, J. R. Ruzycki, R. W. Carmichael, J. Haire, J. Schricker. 2008 Escapement and Productivity of Spring Chinook and Summer Steelhead in the John Day River Basin", 2006-2007 Technical Report, project no. 199801600, 98 electronic pages (BPA Report DOE/BP-00036462-1).

Winter, J. D. 1996. Advances in underwater biotelemetry. In: Murphy, B. R., and D. W. Willis, eds. Fisheries techniques, 2nd edn. Bethesda, MD: American Fisheries Society, pp. 555-590.

Wissmar, R. C., J. E. Smith, B. A. McIntosh, H. W. Li, G. H. Reeves, and J. R. Sedell. 1994. A history of resource use and disturbance in riverine basins of eastern Oregon and Washington (early 1800s-1990s). Northwest Science 68:1-35.

EFFECTIVENESS OF FLOW MANAGEMENT AND RAINBOW TROUT HARVEST ON LONG-TERM VIABILITY OF NATIVE YELLOWSTONE CUTTHROAT TROUT IN THE SOUTH FORK SNAKE RIVER

Laurie Battle[1], Rob Van Kirk[2], and Bill Schrader[3]

[1]Associate Professor, Department of Mathematical Sciences, Montana Tech, 1300 W. Park St., Butte, MT, 59701

[2]Associate Professor, Department of Mathematics, Humboldt State University, 1 Harpst St., Arcata, CA 95521

[3]Fisheries Biologist, Idaho Department of Fish and Game, Nampa, ID

ABSTRACT — The South Fork Snake River supports one of the last remaining large-river populations of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (YCT). Rainbow trout *O. mykiss* and rainbow × cutthroat hybrid trout (collectively, RHT) established a self-sustaining population there in the mid-1980s. Since 2001, all RHT entering tributaries to spawn have been removed to ensure hybridization does not occur in tributaries, and we assume that this will continue. In 2004, U.S. Bureau of Reclamation began delivering a spring "freshet" from Palisades Dam, and Idaho Department of Fish and Game removed harvest limits on RHT. We evaluated effectiveness of these management actions with a stochastic simulation model parameterized with observed data. The model, which incorporates hybridization in the main river and competition among all juveniles, explains observed RHT invasion rates. Higher winter flow increases total RHT + YCT recruitment, and higher freshet flow decreases RHT recruitment. Our results suggest that increased percentage and abundance of YCT since 2004 has resulted from implementation of the freshet and harvest programs. About 20% RHT exploitation is required to maintain YCT in equal abundance with RHT. Increased percentage of YCT requires higher RHT harvest, higher maximum flows, or both and increased abundance of YCT requires higher winter flows.

INTRODUCTION

The South Fork Snake River downstream of Palisades Reservoir in eastern Idaho supports a world-renowned recreational fishery and one of the last remaining large-river populations of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (YCT) in its historic range. Nonnative rainbow trout *O. mykiss* and rainbow × cutthroat hybrid trout (collectively, RHT) established a self-sustaining population in the upper South Fork in the mid-1980s. Since that time, the RHT population has grown steadily while the YCT population has declined. Nonnative brown trout *Salmo trutta* are also present in the upper South Fork, but their population has been stable in low abundances for at least half a century. In an effort to control the RHT invasion, the Idaho Department of Fish and Game (IDFG) has implemented a three-pronged management approach: operation of weirs to prevent RHT from entering YCT spawning tributaries, delivering a spring-time freshet from Palisades Dam that limits survival of RHT eggs and fry, and encouraging unlimited angler harvest of RHT (Fredericks et al. 2004). Angler harvest has been used as a means of controlling nonnative species in other areas with some success (Hansen et al. 2010; Stapp and Hayward 2002). A simple Lotka-Volterra model for interaction of YCT and RHT in the study area showed that hybridization is the predominant mechanism for RHT invasion and competition is likely a contributing factor (Van Kirk et al. 2010). In this paper, we use simulation modeling to assess the effectiveness of angler harvest and flow management in controlling the RHT invasion and maintaining a viable YCT population, assuming that removal of RHT from the tributaries continues.

METHODS

We used a stochastic, age-structured simulation model to predict population trends in a 4.9-km study reach of the upper South Fork Snake River under various management scenarios. We considered rainbow trout and rainbow × cutthroat hybrid trout as a single population (RHT) due to their similar behavioral patterns in terms of spawning and aggression, and we designated two distinct subpopulations of YCT—tributary-spawning and river-spawning—which differ in their interaction with RHT (Van Kirk et al. 2010). Interaction between RHT and YCT in the model occurs via hybridization (main river only) and via competition among age-0 individuals.

Demographic Parameters

With the exception of egg-to-fry survival, all demographic parameters were estimated from samples of fish collected in and adjacent to the study reach between 1996 and 2003 (Table 1). Maximum life expectancy *m* was taken to be the oldest fish in the sample, as determined from otoliths. Adult survival was assumed to be constant across age classes and was estimated for each year using Heincke's estimator $S_A = (N - N_1)/N$ (Ricker 1975), where *N* is total population size (age classes 1 through *m*) and N_1 is abundance of age-1 fish. Mean fry production per individual of age *j* is given by $R_j = 0.5P_{mature}(L_j) \times F(L_j) \times S_{egg-fry}$, where L_j is the mean length of individuals of age class *j*, $P_{mature}(L_j)$ is the probability that a female of length L_j is reproductively mature, $F(L_j)$ is the fecundity of a mature female of length L_j, $S_{egg-fry}$ is the survival of individuals from egg to fry swim-up stage, and the factor of 0.5 accounts for a 1:1 sex ratio. We did not consider the proportion of females in samples of fish from the study reach (Table 1) sufficiently different from 50% for modeling purposes. Mean length-at-age was estimated from a von Bertalanffy growth curve fit to otolith-determined ages using nonlinear, least-squares regression. This curve was used to estimate mean annual growth increment for each age class and inter-annual variance in growth increments. Maturity probabilities were estimated with logistic regression. Dependence of fecundity on length was estimated from linear regression of $\log_e$(eggs) versus $\log_e$(length). Egg-to-fry survival has not been estimated in the study area, so we used the estimates for YCT in a neighboring watershed (Koenig 2006). Adult survival, egg-to-fry survival, and growth increments were selected randomly each year from normal distributions. Demographic stochasticity was not included in the model. When included in particular simulation runs, harvest was modeled by multiplication of the number of catchable size RHT by the harvest exploitation rate.

Table 1. Demographic parameters and functional relationships used in the simulation model. Sample sizes, *n*, are reported in parentheses, where applicable.

	Cutthroat trout	Rainbow/hybrid trout
Maximum life expectancy, *m* (years)	10 (*n* = 184 fish)	6 (*n* = 156 fish)
Mean adult survival, $\bar{S}_A$	0.660 (*n* = 16 years)	0.379 (*n* = 13 years)
Length (mm) of individuals of age *j*, L_j	$L_j = 460[1 - \exp\{-0.293(j + 0.625)\}]$ (*n* = 184 fish)	$L_j = 657[1 - \exp\{-0.217(j + 0.924)\}]$ (*n* = 156 fish)
Maturity probability, $P_{mature}(L_j)$	$P_{mature}(L_j) = \frac{1}{1 + \exp\{12.9 - 0.0397L_j\}}$ (*n* = 165 fish)	$P_{mature}(L_j) = \frac{1}{1 + \exp\{12.0 - 0.0307L_j\}}$ (*n* = 191 fish)
Fecundity, $F(L_j)$	$F(L) = 7.37 \times 10^{-6} L_j^{3.22}$ (*n* = 30 fish)	$F(L) = 4.33 \times 10^{-4} L_j^{2.50}$ (*n* = 65 fish)
Egg-to-fry survival, $S_{egg-fry}$	0.22	0.22
Females in adult population	53% (*n* = 312 fish)	45% (*n* = 420 fish)

Flow-Recruitment Relationships

The effects of stream-flow on trout recruitment in the Rocky Mountains are well documented. High spring-time flows during the snowmelt freshet can displace RHT fry and eggs (Nehring and Anderson 1993; Fausch et al. 2001), while native YCT generally spawn after this freshet and are therefore not harmed by it. We modeled RHT fry survival as a function of maximum freshet flow with logit regression of data from the study reach (Figure 1). This survivorship multiplies egg-to-fry survival for RHT, producing an additional mortality factor based on the maximum magnitude of the freshet flow. Availability of concealment habitat determines over-winter survival of age-0 individuals of both species (Griffith and Smith 1993; Mitro et al. 2003) and hence the carrying capacity for the cohort of individuals surviving to age 1. We used weighted nonlinear regression to estimate this total age-1 carrying capacity as a function of winter flow from data collected in the study reach (Figure 1). Winter flow and maximum flow were randomly selected each year from log-normal distributions with variance, covariance, and temporal autocorrelation as estimated from observed flow data.

Competition Model

To incorporate the effect of species competition, we constructed a model in which all young-of year compete between their emergence as fry and the end of winter. Pairs of fish compete in a series of "rounds" in which the loser dies or emigrates and the winner advances to the next round. Assuming that individuals encounter each other randomly, the species composition distribution of these pairs occurs in proportion to the abundance of each species in that round. Let be the RHT young-of-year abundance and be the YCT young-of-year abundance at the beginning of round *j*. In round *j*, the fraction of pairs in which both fish are RHT is

, and the fraction of pairs in which one fish is RHT and the other is YCT is

. Then

, where is the proportion of RHT-YCT competitions won by the RHT. The parameter was taken to be 0.78 based on an experimental study of competitive interactions between RHT and YCT (Seiler and Keeley 2007). The value is the difference between and the total abundance in round

Rounds of pair-wise competitions continue until the total number of fish remaining equals the age-1 carrying capacity, which is determined from the randomly selected winter flow for that particular year and the relationship depicted in Figure 1.

Hybridization Model

We assumed that hybridization occurs only between RHT and the river-spawning YCT subpopulation and occurs in proportion to temporal overlap in observed spawn timing distributions between female YCT and male RHT (Figure 2). We assumed that individuals spend one week in the spawning area and that eggs are fertilized randomly by males in proportion to their relative abundance.

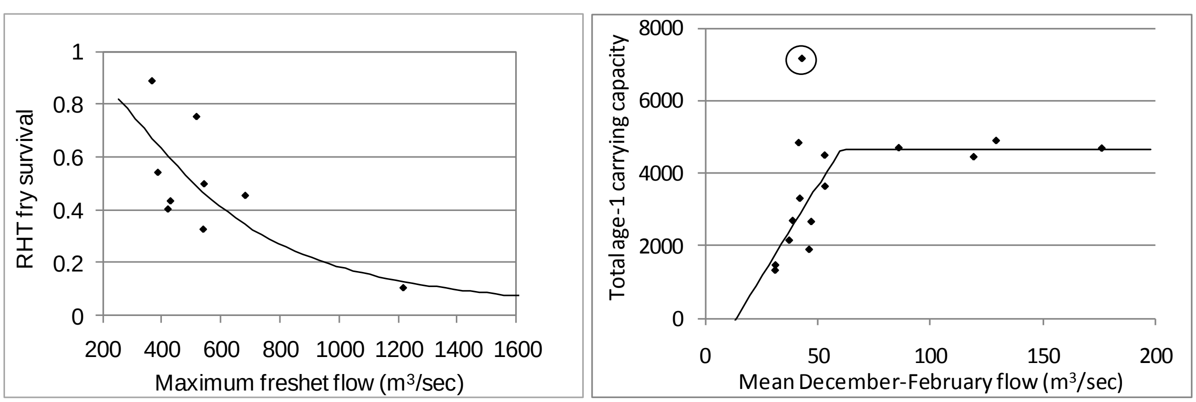


Figure 1. Dependence of juvenile population parameters on maximum freshet flow (left) and mean winter flow (right). Circled data point in right panel was considered an outlier and omitted prior to curve-fitting.

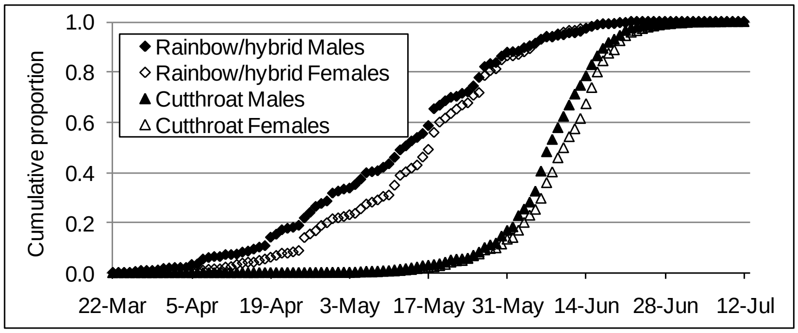


Figure 2. Distribution of observed spawning times of Yellowstone cutthroat trout and rainbow and rainbow x cutthroat hybrid trout.

Let be the fraction of female YCT that spawn in week *i*, be the fraction of male RHT that spawn in week *i*, and be the fraction of male YCT that spawn in week *i*. If is the total number of male RHT spawners, is the total number of male river-spawning YCT, and is the number of eggs laid by all river-spawning YCT, then the number of YCT eggs fertilized by male RHT is ,

where *i* ranges over all weeks of spawning activity. All such eggs are subtracted from the YCT egg total and added to the RHT egg total.

Stochastic Simulation and Analysis Procedures

To assess model performance in predicting observed population trends since RHT were first detected in 1989, 100 independent simulations were initiated with the population abundances observed in that year (310 RHT, 7,890 YCT), assuming that tributary spawners comprised 40% of the total YCT population in 1989 (Van Kirk et al. 2010). These simulations were run through 2008, and mean population values for each year were compared with those observed. Flow and harvest values were fixed each year at observed values, so that the only stochastic elements in these simulations were growth and survival. To model a "no action" scenario, we performed a set of 100, 100-year simulations initiated at 1989 values using flow distribution parameters as observed over the period 1988-2003 (post-RHT invasion but pre-freshet management) and no RHT harvest. The anomalous year of 1997 was omitted in determining the flow parameters for this period; a rain-on-snow event that year produced runoff that exceeded the reservoir capacity and resulted in a one-day maximum flow of 1,215 m^3/sec. Lastly, we initiated a set of 820 simulations at the population values observed in 2003 (4,114 YCT and 4,653 RHT), and ran these simulations for 25 years under management scenarios that included values of freshet flow, winter flow, and harvest that we considered to be feasible under current regulatory and sociological conditions. We performed simulations at RHT harvest exploitation rates of 0, 10%, 20%, and 30%, and at each of these, under winter flow distribution means of 22.7, 34.0, 45.3, 56.6, and 68.0 m^3/sec. At each of these exploitation × winter flow combinations, we performed simulations at 41 values of maximum freshet flow distribution means ranging from 396.5 m^3/sec to 623.0 m^3/sec, inclusive, in steps of 5.7 m^3/sec. For each 25-year simulation, we recorded the final YCT and RHT population values and the mean winter and freshet flow values. Because of stochasticity, these values deviated somewhat from the specified distributional means. To facilitate interpretation of the simulation output, we performed linear regression of YCT abundance and percent YCT in the final simulation year as functions of mean freshet flow, mean winter flow, harvest exploitation, and the interaction of harvest with each of the flow variables. The abundance data were square-root transformed to meet residual assumptions.

Results

The model performed well in predicting the observed invasion of RHT; the mean of 100 stochastic simulations explained 71.9% of total variability in observed RHT abundance over the period 1990-2008 (n = 15, Figure 3). The model also performed well in predicting general trends in the YCT population, although the mean explained only 44.4% of total variability in observed YCT abundance (Figure 3).

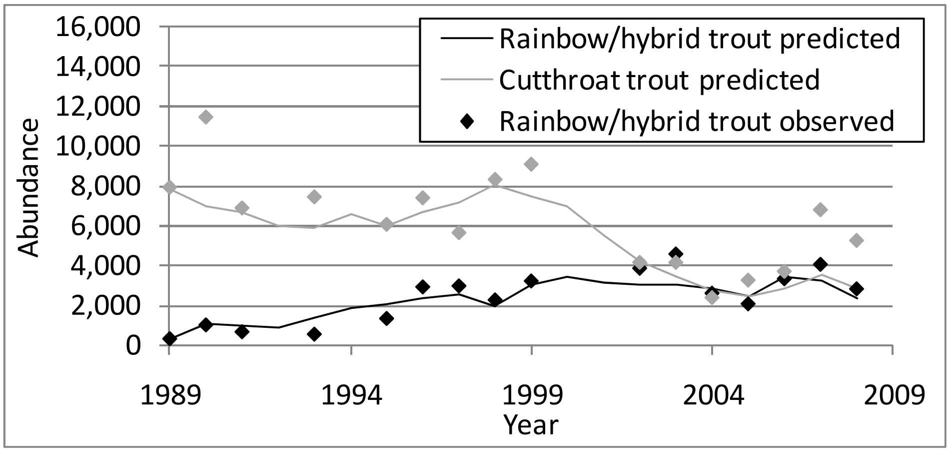


Figure 3. Observed and model-predicted (mean of 100 stochastic simulations) population size of Yellowstone cutthroat trout and rainbow and rainbow x cutthroat hybrid trout .

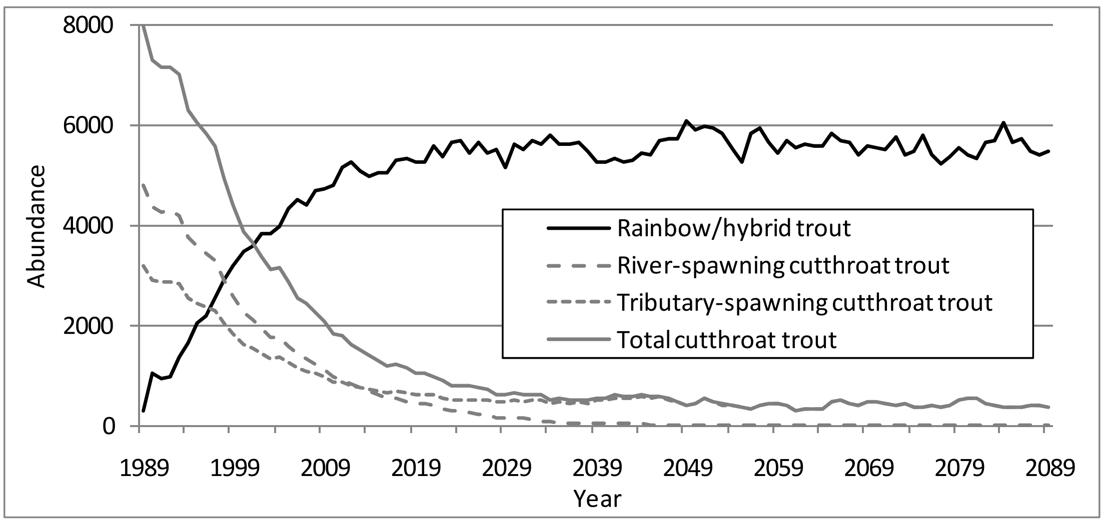


Figure 4. Model-predicted population trends (mean of 100 stochastic simulations) of Yellowstone cutthroat trout and rainbow and rainbow x cutthroat hybrid trout under removal of rainbow and hybrid trout from tributaries but no flow management (1988-2003 hydrologic conditions) or harvest.

However, a 95% percent prediction interval for modeled abundances contained all observed population values except the YCT value for 1990. Under observed 1988-2003 flow conditions and a no-harvest scenario, species composition reached equilibrium in about 50 years, after which time the river-spawning subpopulation of YCT was lost and abundances averaged about 5,600 RHT and 440 YCT in the 4.9-km study reach (Figure 4).

Abundance of YCT after 25 years of management was an increasing function of freshet flow, winter flow and RHT harvest exploitation (Figure 5). The fitted linear regression equation ($R^2 = 0.958$, df = 814) was

$$\sqrt{YCT} = -32.1 + 0.979H + 0.0968Q_F + 0.193Q_W - 0.000713H * Q_F + 0.0165H * Q_W \ , \quad (1)$$

where YCT = YCT abundance, H = RHT harvest exploitation (percent), Q_F = maximum freshet flow

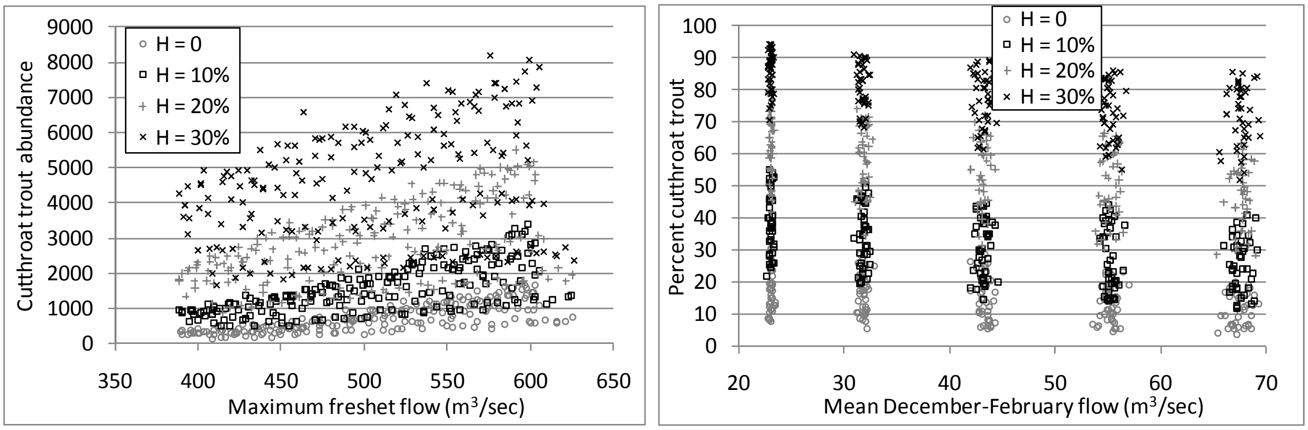


Figure 5. Predicted cutthroat trout abundance after 25 years as a function of maximum freshet flow (left), mean winter flow (right), and rainbow/hybrid harvest exploitation, H.

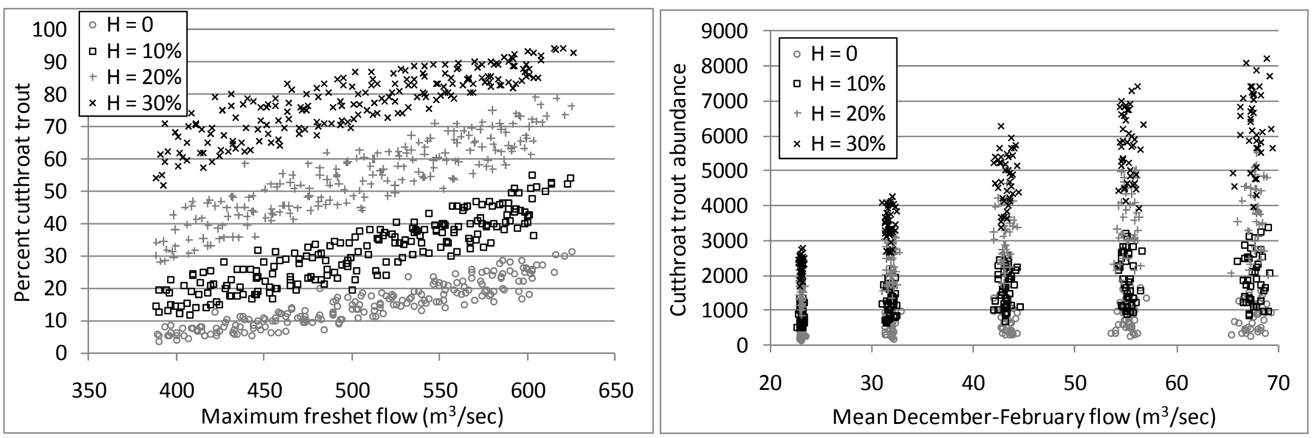


Figure 6. Predicted percentage of cutthroat trout in the *Oncorhynchus* population after 25 years as a function of maximum freshet flow (left), mean winter flow (right), and rainbow/hybrid harvest exploitation, H.

(m^3/sec), and Q_W = winter flow (m^3/sec). Percentage of YCT after 25 years was an increasing function of freshet flow and RHT harvest but a decreasing function of winter flow (Figure 6). The fitted regression equation (R^2 = 0.993, df = 814) was

$$\%YCT = -36.8 + 1.850H + 0.111Q_F - 0.155Q_W + 0.000964H * Q_F - 0.00497H * Q_W \,. \quad (2)$$

Discussion

Our simulation results suggest that in absence of flow and harvest management, the tributary removal program alone is not sufficient to prevent nearly complete displacement of the YCT population by RHT. Only a small, tributary-spawning component of the YCT population is predicted to persist under these conditions, and this assumes that tributary-spawned YCT exhibit complete life-history fidelity by returning to their natal tributaries to spawn. The model predicted that without implementation of the freshet and harvest programs in 2004, YCT would have comprised only about 33% of the total *Oncorhynchus* population in 2008 (Figure 4), compared with 54% observed (Figure 3). Numerous factors—including carryover storage and irrigation demand in the entire upper Snake River irrigation system, snowpack, spring-time weather, and flood control rules—constrain the ability of the U.S. Bureau of Reclamation to deliver a freshet flow of optimal magnitude and timing from Palisades Dam in any given year. Despite these constraints, over the first 7 years of the freshet program, peak flow during the optimal late May-early June period reached the target of 538 m^3/sec in five of those years, resulting in a 7-year mean of 511 m^3/sec, compared with a mean of 498 m^3/sec over the period 1988-2003 (excluding the flood of 1997). Concurrently, angler harvest of RHT increased from 7% exploitation in 2003 to 20% in 2005. Thus, our results provide evidence that the flow and harvest components of the three-prong management approach have begun to slow the RHT invasion and allow YCT numbers to rebound.

Assuming that freshet management continues as it has since 2004 and that winter flows continue to average 43 m^3/sec, the model predicts that a 20% exploitation rate will result in a population of 2,720 (56%) YCT and 2,150 (44%) RHT in the year 2028. Improving the species composition to 75% YCT could be achieved by increasing the freshet mean to 625 m^3/sec under 20% harvest exploitation, increasing harvest exploitation to 30% under the current freshet regime, or some intermediate increase in both variables (Figure 6). Although current water management constraints make it unlikely that the freshet mean will be as high as 625 m^3/sec, this flow is only slightly lower in magnitude than the river's natural (pre-dam) bank-full discharge, and thus attainment of this flow in at least some years will benefit not only YCT recovery but also maintenance of in-channel, floodplain, and riparian habitat throughout the South Fork downstream of Palisades Dam (Hauer et al. 2004).

Winter flow management is not currently a formal component of the three-prong approach, but the simulation results show that winter flow magnitude exerts a substantial influence over YCT abundance and percentage. Decreased winter flow decreases YCT abundance, but it increases the long-term fraction of YCT in the population. This occurs because age-0 survival of both species is negatively affected by low winter flows via lower availability of concealment habitat, but because RHT have a lower annual survival rate and shorter life expectancy, the RHT population is more sensitive to occurrence of years in which recruitment of age-1 fish is low. Because the YCT population is more resilient to low recruitment, the long-term population of YCT that can be supported in the study reach is higher than that of RHT (Van Kirk et al. 2010). Thus, maintaining a higher fraction of YCT in the population provides higher overall abundance of *Oncorhynchus* for anglers at any given level of recruitment allowed by winter flow availability. In some years, it may be necessary to store extra water in Palisades Reservoir during the winter (decrease winter flows downstream) to increase the likelihood of a favorable freshet peak the following spring. In these years, the decrease in abundance of YCT that occurs due to decreased winter flows is balanced by the increase in YCT percentage that results from both decreased winter flows and the increased freshet flow made possible by storing the extra water. In the range of flow values we considered (those currently feasible), the interaction terms in the regression equations show that increased harvest increases the sensitivity of YCT percentage to both winter flow and freshet flow. Thus, the harvest and freshet programs each increase the effectiveness of the other in increasing the fraction of YCT in the population. Increased harvest also increases the incidental positive effect of lower winter flows on species composition. On the other hand, increased harvest decreases the sensitivity of YCT abundance to freshet flow and increases its sensitivity to winter flow. Although our results show that winter flow is an important determinant of the long-term trajectory of the YCT population, more careful analysis of the complex relationships among species interaction, population abundance, hybridization, competition, winter flow, freshet flow and harvest is needed to develop strategies for managing winter flow for optimal benefit to YCT and the fishery.

Our results suggest that the harvest and freshet flow programs have slowed the RHT invasion and allowed recovery of the YCT population in the South Fork Snake River. Water managers and anglers have both shown willingness to participate in the management effort, as evidenced by increases in mean freshet flow magnitude and RHT harvest. Further increases in RHT harvest are desirable to offset the higher RHT fry survival rates that will inevitably occur in years when freshet flow objectives cannot be met due to water management constraints. On the other hand, increased freshet peaks, when possible, will not only increase the effectiveness of the harvest program but will also benefit the entire river and riparian ecosystem. Sustainability of YCT and the multi-million dollar recreational fishery as a whole will require addition of a winter flow component to the three-prong approach and a more thorough understanding of the complex relationships among species interactions, flow, and harvest.

Literature Cited

Fausch, K., Y. Taniguchi, S. Nakano, G.D. Grossman, and C.R. Townsend. 2001. Flood disturbance regimes influence rainbow trout invasion success among five holarctic regions. Ecological Applications 11:1438-1455.

Fredericks, J., B. Schrader, and R. Van Kirk. 2004. A collaborative, multi-faceted approach to Yellowstone cutthroat trout conservation in the South Fork of the Snake River, Idaho. Pages 158-166 in Moore, S.E.; Carline, R.F.; Dillon, J., eds. 2004. Working together to ensure the future of wild trout; proceedings of Wild Trout VIII symposium—30th anniversary; 2004 September 20-22; Yellowstone National Park, Wyoming.

Hansen, M.J., D. Schill, J. Fredericks, and A. Dux. 2010. Salmonid predator-prey dynamics in Lake Pend Oreille, Idaho, USA. Hydrobiologia 650:85-100.

Hauer, F.R., Lorang, M.S., Whited, D., and Matson, P. 2004. Ecologically Based Systems Management: The Snake River—Palisades Dam to Henrys Fork. Open File Report 183-04, Flathead Biological Station, Polson, Montana.

Koenig, M.K. 2006. Habitat and biotic factors influencing distribution and recruitment of juvenile cutthroat trout in the Teton River, Idaho. Master's thesis. Utah State University, Logan.

Mitro, M.G., A.V. Zale, and B.A. Rich. 2003. The relation between age-0 rainbow trout (*Oncorhynchus mykiss*) abundance and winter discharge in a regulated river. Canadian Journal of Fisheries and Aquatic Sciences 60:135-139.

Nehring, R.B., and R.M. Anderson. 1993. Determination of population-limiting critical salmonid habitats in Colorado streams using the physical habitat simulation system. Rivers 4:1-19.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada 191:1-382.

Seiler, S.M., and E.R. Keeley. 2007. A comparison of aggressive and foraging behavior between juvenile cutthroat trout, rainbow trout and F1 hybrids. Animal Behavior 74:1805-1812.

Stapp, P., and G.D. Hayward. 2002. Effects of an introduced piscivore on native trout: insights from a demographic model. Biological Invasions 4:299-316.

Van Kirk, R.W., L. Battle, and W.C. Schrader. 2010. Modelling competition and hybridization between native cutthroat trout and nonnative rainbow and hybrid trout. Journal of Biological Dynamics 4:158-175.

YELLOWSTONE CUTTHROAT TROUT CONSERVATION EFFORTS ON THE SOUTH FORK SNAKE RIVER

B. High

Regional Fisheries Biologist, Idaho Department of Fish and Game, Idaho Falls, Idaho

ABSTRACT — The South Fork Snake River supports a healthy population of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (YCT). However, the YCT fishery is threatened by nonnative rainbow trout *O. mykiss* (RBT) that compete and hybridize with YCT. A three-pronged management approach was implemented in 2004. The three prongs include maintaining YCT strongholds in tributaries, managing flow in the main river, and increasing angler harvest of RBT. Weirs have been installed and maintained on four major spawning tributaries of the South Fork of the Snake River for the last decade. Each spring, adult trout are sorted and only YCT are allowed upstream. However, research has shown that tributary strongholds alone cannot protect YCT indefinitely. Flow management during spring with appropriate timing and magnitude can impede RBT spawning success while benefitting YCT success and is a necessary component of the management program. However, ideal spring flows cannot always be realized due to irrigation, snowpack, and reservoir storage. The final component, RBT harvest, is most easily implemented. Anglers play a key role in conservation efforts in concert with tributary and flow management. Since 2004, conservation efforts have experienced success and setbacks. Yellowstone cutthroat trout populations in the South Fork will continue to be a conservation-reliant species.

INTRODUCTION

The native trout of the South Fork Snake River is the Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*. This population is one of the few remaining healthy fluvial populations within their historical range in Idaho (Thurow et al. 1988; Van Kirk and Benjamin 2001; Meyer et al. 2006). Across the majority of the species' range, Yellowstone cutthroat trout (YCT) have experienced dramatic reductions in abundance and distribution. In August 1998, conservation groups petitioned the United States Fish and Wildlife Service (USFWS) to list Yellowstone cutthroat trout under the Endangered Species Act (ESA). In February 2001, the listing petition was denied, and the conservation groups filed a lawsuit in January 2004, which led to a 12-month review of the current status of YCT. The USFWS determined that YCT did not warrant listing under the ESA in February 2006 (USFWS 2006).

The South Fork Snake River YCT population is both ecologically and economically important. Yellowstone cutthroat trout are widely known among anglers as a trout that are easy to catch (Griffith 1993; Thurow et al. 1988) and readily take dry flies. A substantial fishery has been established on the YCT of the South Fork Snake River. The South Fork Snake River fishery generated approximately US $12 million in local income and supported an estimated 341 jobs in 2004 (Loomis 2005).

While the YCT population in the South Fork Snake River is considered healthy (Meyer et al. 2006), recent abundance estimates have declined (Moore and Schill 1984) due to many factors including hybridization with nonnative rainbow trout *O. mykiss*. Hybridization has been blamed for YCT declines across their native range (Krueger and May 1991; Leary et al. 1995). Yellowstone cutthroat trout and rainbow trout (RBT) are closely related taxonomically and have similar life histories. Rainbow trout spawn slightly before YCT, which typically spawn during the descending limb of the spring hydrograph peak. However, spatial and temporal overlap of RBT and YCT spawning has been documented in the main river and spawning tributaries of the South Fork Snake River (Henderson et al. 2000). Currently, YCT and RBT select similar spawning habitat in the main river (side channels), and in tributaries RBT tend to spawn in the lower reaches while YCT migrate further upstream (Henderson et al. 2000; Meyer et al. 2006).

Nonnative RBT have been stocked in the South Fork Snake River from the 1880s through 1981, but comprised only a small portion of the species composition (Moore and Schill 1984). Through 1981, catchable-sized fertile RBT were stocked at various locations including Palisades Creek, Rainey Creek, and a wild run of RBT had become established in Palisades Creek by 1982 (Moore and Schill 1984). Densities of RBT remained low through the early 1990s (Schrader et al. 2002), but increased in abundance with a concurrent decline in YCT from the mid-1990s through 2003 (Garren et al. 2006). Rainbow x cutthroat hybrid trout (hereafter included with RBT) abundances have also increased. The threat of hybridization to YCT populations is substantial (Clancy 1988; Thurow et al. 1988), because females from both YCT and RBT hybridize and their offspring are fertile (Henderson et al. 2000).

Spring flows resembling a natural hydrograph for a snowmelt-driven system benefits YCT while adversely affecting rainbow trout (Moller and Van Kirk 2003). The impacts of this natural-shaped hydrograph are more pronounced after low winter flows. Palisades Reservoir, as a flood control and irrigation storage reservoir, alters flows downstream by increasing late winter flows to make room for spring runoff, thereby diminishing the spring run-off peak downstream and increasing the summer flows. In other words, the altered hydrograph tends to be smoothed relative to the natural snowmelt hydrograph. Maximum daily spring flows out of Palisades Reservoir from 1999 to 2003 averaged 355 cms. Moller and Van Kirk (2003) recommend a natural-shaped spring peak hydrograph that is at least 15 times higher than the prior winter's minimum flow to improve YCT recruitment relative to rainbow trout (RBT) in the South Fork Snake River.

A change in the South Fork Snake River fishery management was implemented in 2004 to address the increasing RBT population and decreasing YCT population. The current management goal is to ensure YCT are the dominant trout in the South Fork Snake River (IDFG 2007), and the approach is three-pronged. The first component deals with reducing RBT abundance through harvest, the second component involves maintaining tributaries as YCT spawning strongholds, and the third component deals with altering flows to benefit YCT.

Study Area

The Snake River originates in Yellowstone National Park and flows south through Grand Teton National Park and the Jackson Hole valley before turning west and flowing into Palisades Reservoir at the Idaho – Wyoming state line. The 106-km portion of the Snake River that runs from Palisades Dam to the confluence with the Henrys Fork is commonly referred to as the South Fork Snake River. Anglers and biologists divide the South Fork Snake River into three segments. The first segment, called the upper river, runs from Palisades Dam to Pine Creek through a relatively unconfined valley. The first 13 km of the upper river downstream of the dam is a simple channel. From this point, the river braids around numerous islands. All but one of the four main YCT spawning tributaries enter the South Fork Snake River in this upper river, including Palisades Creek, Rainey Creek, and Pine Creek (Figure 1). The second segment of the South Fork Snake River runs from Pine Creek downstream to Heise, and is commonly referred to as the Canyon. Burns Creek, the fourth major YCT spawning tributary enters the South Fork in the Canyon. The last segment of the South Fork Snake River runs from Heise to the confluence with the Henrys Fork, and is commonly referred to as the lower river. There are no major YCT spawning tributaries in the lower river, and while constant water temperatures from Palisades Dam moderate winter conditions in the upper and canyon sections, winter conditions in the lower river are usually more severe than upstream (Moller and Van Kirk 2003). The Conant and Lorenzo monitoring reaches of the South Fork are in the upper river and lower river sections, respectively.

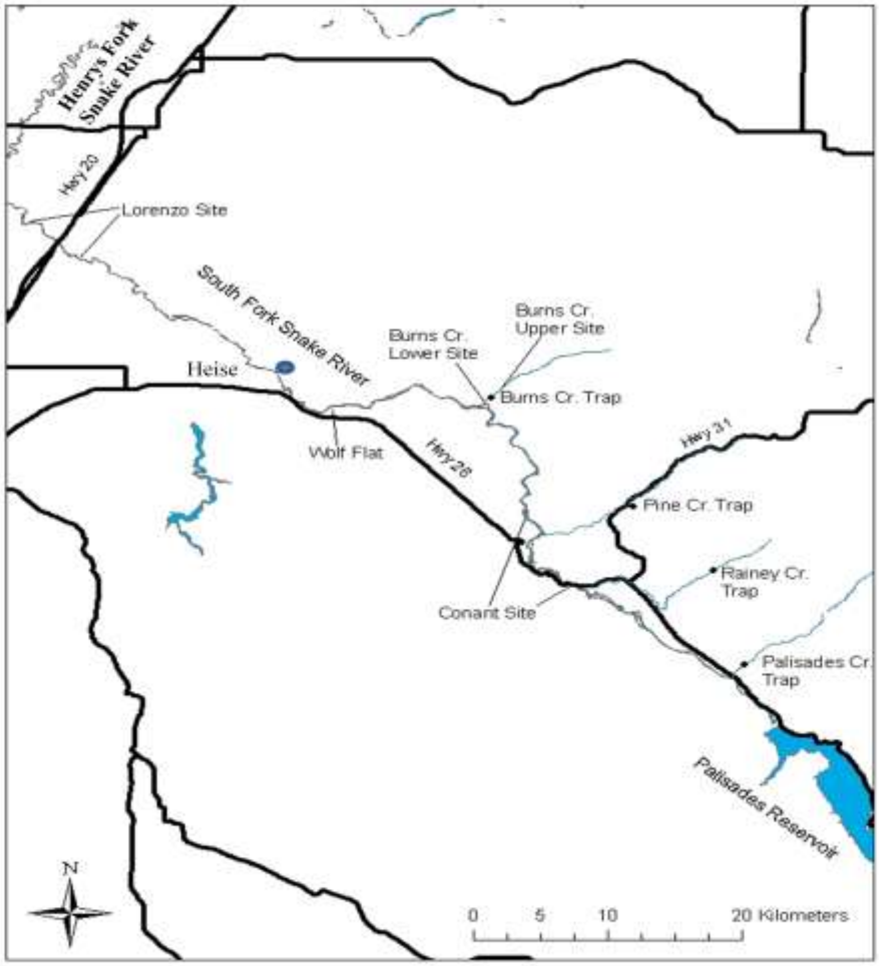


Figure 1. The South Fork Snake River, spawning tributaries, and monitoring locations in southeast Idaho.

Methods

Weirs

Migration barriers and traps have been operated on the four main spawning tributaries of the South Fork Snake River since 2001. Various weir types have been used to maximize trapping efficiency, including floating panel, Mitsubishi, hard picket, electric, and fall-velocity weirs. Weirs and traps are typically installed in mid-March to mid-April and operated through mid-July.

All fish captured at the weirs were visually identified to species, sexed according to expression of milt or eggs and head morphology, and measured to the nearest mm (total length). Yellowstone cutthroat trout were released upstream of the weir. Rainbow trout and rainbow x cutthroat trout hybrids were removed from the run and transported to a nearby kids fishing pond. Weir efficiencies were estimated in some years by marking all YCT with a fin clip or caudal punch and later recapturing fluvial YCT upstream of the fish trap via electrofishing or a downstream migrant trap. Weir efficiencies were estimated by dividing the number of marked fluvial YCT by the total number of YCT captured that were as large as fluvial YCT (>350 mm).

Exploitation

We estimated the annual exploitation rate of rainbow trout in the South Fork Snake River in 2003 and 2005 using roving creel surveys (see Schrader and Fredericks 2006a; Schrader and Fredericks 2006b). Non-reward tag return data from rainbow trout marked with anchor tags were used to estimate annual exploitation of RBT in 2006, 2007, and 2009. Exploitation rates calculated using tag return data were adjusted for tag loss, tagging mortality, and volunteer reporting rate using correction values specific to the South Fork Snake River as described by Meyer et al. (2008).

South Fork Snake River Monitoring

We sampled the Lorenzo and Conant monitoring reaches of the South Fork Snake River during each fall to estimate abundance of trout. Estimates were calculated for each species and only included age-1 and older trout (Schrader and Fredericks 2006a). We used a jet boat outfitted with electrofishing gear and pulsed DC electrical current to capture fish. At each monitoring reach, we typically collected trout over 2 d for marking followed by a 7-d rest and a 2-d recapture effort. The Lorenzo monitoring reach is usually sampled during the last 2 weeks of September while the Conant monitoring reach is sampled during the first 2 weeks of October. We attempted to capture all trout encountered. Trout were identified to species and measured to the nearest mm (total length). Fish that were large enough to be included in population estimates were captured during the marking runs and marked with a hole punch in the caudal fin for identification during recapture runs. Abundance estimates and 95% confidence intervals were calculated using the log-likelihood method in Montana Department of Fish Wildlife and Park's software Mark/Recapture for windows or MR 5.0 (MFWP 1997). The log-likelihood method adjusts estimates based on modeled capture efficiencies, which differ by fish size (Schill 1992). Abundance estimates were standardized to fish per kilometer for comparison with estimates collected since 1986. As with most mark - recapture surveys, we assumed the population was closed, the probability of capture was the same for individual fish for each run, fish did not lose their marks, marked fish mixed randomly with unmarked fish, and the marks were recognized properly (Ricker 1975).

We monitor YCT and RBT recruitment to gauge the effectiveness of managed spring flows aimed to benefit YCT. We cannot effectively sample age-0 fish; therefore, we use the abundance of age 1 YCT and RBT to evaluate the effects of spring flows during the previous year.

Results

Weirs

At Burns Creek, we have captured as many as 3,156 YCT and 46 RBT between 2001 and 2010 (Table 1). Trap efficiencies did not exceed 36% when floating panel and Mitsubishi weirs were used, but have exceeded 98% for both years that the fall-velocity barrier has been in operation. At Pine Creek, we have captured as many as 2,852 YCT and 40 RBT with weir efficiencies between 20 and 98%. We have captured up to 142 YCT and 3 RBT at the Rainey Creek weir. We could not estimate weir efficiencies at Rainey Creek due to low numbers of spawning YCT. We have captured as many as 1,072 YCT and 310 RBT at the Palisades Creek weir with efficiencies ranging from 10 to 98%.

Table 1. Summary tributary fish trap operation dates, efficiencies and catches from 2001 through 2010.

				Catch		
Location/ year	**Weir type**	**Operation dates**	**Efficiency (%)**	**Cutthroat trout**	**Rainbow trout**	**Total**
Burns Creek						
2001	Floating panel	Mar. 7 – July 20	16	3,156	3	3,159
2002	Floating panel	Mar. 23 – July 5	--	1,898	46	1,944
2003	Floating panel	Mar. 28 – June 23	17	1,350	1	1,351
2006	Mitsubishi	Apr. 14 – June 30	--	1,539	0	1,539
2009	Fall/velocity	Apr. 9 – July 22	98	1,491	2	1,493
2010	Fall/Velocity	Mar. 26 – July 14	100	1,530	2	1,532
Pine Creek						
2002	Floating panel	Apr. 2 – July 5	--	202	14	216
2003	Floating panel	Mar. 27 – June 12	40	328	7	335
2004	Hard picket	Mar. 25 – June 28	98	2,143	27	2,170
2005	Hard picket	Apr. 6 – June 30	--	2,817	40	2,857
2006	Mitsubishi	Apr. 14 – Apr. 18	--	0	0	0
2007	Mitsubishi	Mar. 24 – June 30	20	481	2	483
2008	Hard picket	Apr. 21 – July 8	--	115	0	115
2009	Hard picket	Apr. 6 – July 15	49	1,356	1	1,357
2010	Electric	Apr. 13 – July 6	--	2,852	2	2,854
Rainey Creek						
2001	Floating panel	Mar. 27 – July 6	--	0	0	0
2002	Floating panel	Mar. 26 – June 27	--	1	0	1
2005	Hard picket	Apr. 7 – June 29	--	25	0	25
2006	Hard picket	Apr. 5 – June 30	--	69	3	72
2007	Hard picket	Mar. 19 – June 30	--	14	0	14
2008	Hard picket	June 19 – July 11	--	14	0	14
2009	Hard picket	Apr. 7 – July 6	--	23	0	23
2010	Hard picket	Apr. 13 – June 29	--	142	0	142
Palisades Creek						
2001	Floating panel	Mar. 7 – July 20	10	491	160	651
2002	Floating panel	Mar. 22 – July 7	--	967	310	1,277
2003	Floating panel	Mar. 24 – June 24	21	529	181	710
2005	Mitsubishi	Mar. 18 – June 30	91	1,071	301	1,372
2006	Mitsubishi	Apr. 4 – June 30	13	336	52	388
2007	Electric	May 1 – July 28	98	737	20	757
2009	Electric	May 12 – July 20	26	202	4	206
2010	Electric	Mar. 19 – July 20	TBD	537	14	551

Exploitation

Annual rates of RBT exploitation on the South Fork Snake River have varied from 13 to 24% between 2003 and 2009 with estimated annual harvests of RBT between 5,070 and 12,322 (Table 2).

South Fork Snake River Monitoring

We have observed high abundances of trout at Lorenzo, where species composition has averaged 23% YCT, 1% RBT, and 76% brown trout *Salmo trutta* (BNT) between 1987 and 2009. Since 2004, our abundance estimates for age-1 and older YCT (≥102) and BNT (≥178) have averaged 136 and 1,070 trout per kilometer, respectively (Figure 2). Abundance estimates for RBT have not been possible with too few marked RBT captured during recapture runs.

At the Conant monitoring site, species composition has averaged 65% YCT, 19% RBT, and 16%, BNT from 1986 through 2009. The estimates for age-1 and older trout per km have averaged 1,341, 544, and 330 for YCT, RBT, and BNT, respectively, over the same time period (Figure 3). The 2009 total trout abundance at Conant (2,541 trout/km) approached our all-time high abundance of 3,013 trout/km that was estimated in 1999. However, over one-half of the total trout abundance at Conant (55%) consisted of rainbow trout. We estimated there were 1,408 age-1 and older rainbow trout/km at the Conant monitoring reach in 2009, which is a significant increase over the 2008 estimate of 574 rainbow trout/km (Figure 3). Most of these rainbow trout were yearling fish, with an estimated density of 1,094 age-1 RBT/km which also was a significant increase. Field observations suggests when the spring freshet peaks in mid- to late May, rainbow trout recruitment is low, but this impact is quickly lessened the later in the year the freshet peaks (Figure 4).

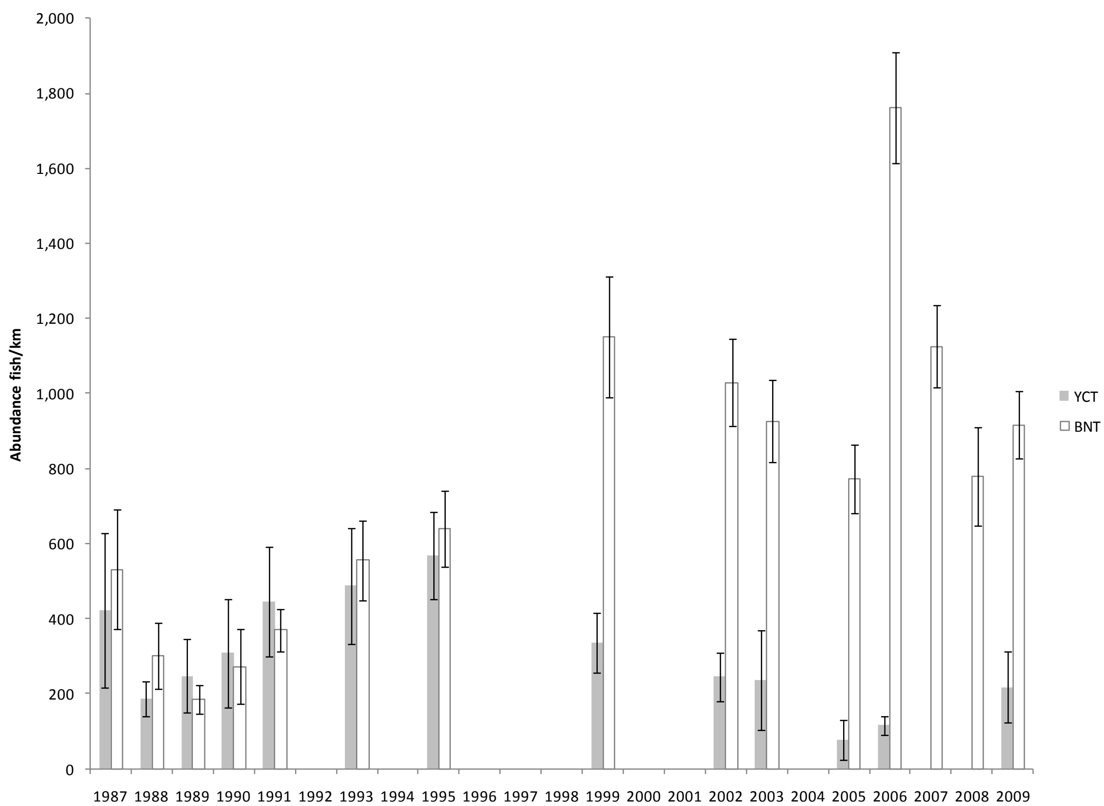


Figure 2. Estimated abundances of Yellowstone cutthroat trout (YCT) brown trout (BNT) at the Lorenzo monitoring site on the South Fork Snake River from 1987 through 2009 with 95% confidence intervals.

Table 2. Summary of the annual exploitation and harvest estimates for rainbow trout (including hybrids) from the South Fork Snake River.

	Annual estimates	
Year	Exploitation (%)	Harvest
2003	12	2,070
2005	21	6,718
2006	24	12,322
2007	14	8,002
2009	13[1]	NA

[1]Partial year (8 month) estimate

Discussion

Weirs

Picket weirs and floating weir designs have proven ineffective at capturing rainbow trout in South Fork Snake River tributaries during years with normal or above average snowpack. Rainbow trout run earlier than Yellowstone cutthroat trout during spring spawning runs in South Fork Snake River tributaries (Henderson et al. 2000; Schrader and Fredericks 2006a), typically when pickets need to be removed to maintain structure integrity during high flows. To address picket weir inefficiencies, we have modified three of the four fish traps. The Burns Creek weir was converted to a fall-velocity barrier and the Palisades and Pine creeks traps were converted to electric barriers. These modifications appear to have substantially increased our trapping efficiencies.

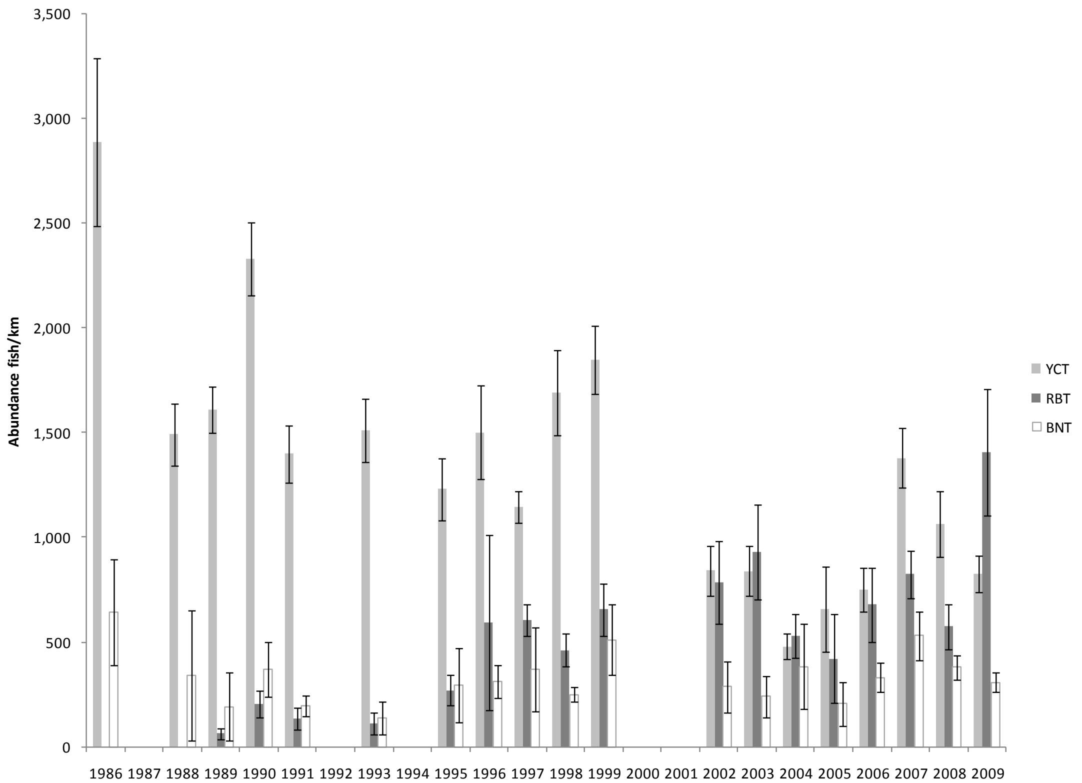


Figure 3. Estimated abundances of Yellowstone cutthroat trout (YCT), rainbow trout (RBT), and brown trout (BNT) at the Conant monitoring site on the South Fork Snake River from 1986 through 2009 with 95% confidence intervals.

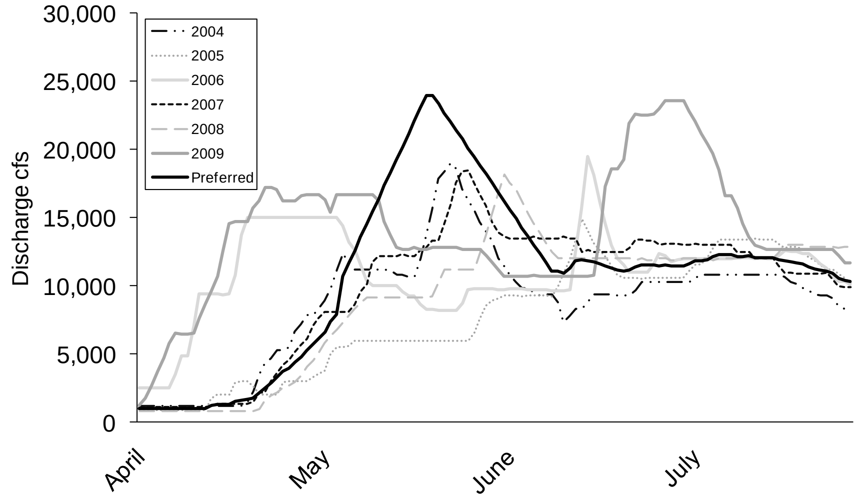


Figure 4. Spring freshet flow discharges from Palisades Dam from 2004 through 2009. Darker lines indicate years when rainbow trout recruitment was low and the preferred spring freshet magnitude and timing and lighter lines indicate years when rainbow trout recruitment was high. The effect of the 2009 flows cannot be evaluated until the fall of 2010.

Effective trapping of rainbow trout is critical to maintain YCT tributary spawning strongholds. While it has been documented that YCT spawn in the main stem of the South Fork Snake River (Henderson et al. 2000), tributaries providing quality spawning habitat without the threat of hybridization with RBT may be more critical for the persistence of YCT in the South Fork Snake River system. Modeling research by Van Kirk et al. (2010) indicates the YCT population in the South Fork Snake River will be difficult to maintain if RBT become dominant in the main stem, regardless if we can maintain reproductive segregation in the tributaries.

Exploitation

While changes in fishing regulations and outreach efforts have increased the exploitation rates of RBT in the South Fork Snake River, there is room for improvement. Despite exploitation rates averaging 20%, the RBT population in the upper river has remained abundant and even increased in abundance significantly between 2008 and 2009 such that for the first time since monitoring started, RBT significantly outnumbered YCT in the Conant monitoring reach. Obviously, 20% annual exploitation is not sufficient to significantly impact the RBT population. Of the three-pronged management approach aimed at conserving YCT, angler harvest is the one with the most opportunity for improvement. Trap modifications have maximized efficiencies for maintaining tributaries as spawning tributaries for YCT, and there is little room for improvement to spring flows that mimic natural hydrograph given irrigation, snowpack, and flood storage constraints. Interestingly, models indicate when RBT exploitation levels exceeding 20% are combined with beneficial spring flows that mimic the natural hydrograph, the benefits to YCT are disproportionally more than when the benefits of similar exploitation and spring flows are modeled separately (Van Kirk et al. 2010).

Non-reward tag return information from 2009 indicates only 50% of the fish that were caught, were harvested. In 2010, IDFG instigated an angler incentive program in attempt to increase the harvest rate of RBT that anglers are already catching. The program includes options for anglers to turn in RBT for potential monetary prizes as well as the ability for non-consumptive anglers to harvest RBT by coordinating the donation of anglers’ catches to the local food bank. The impact of the angler incentive program on RBT exploitation rates in the South Fork

Snake River will be evaluated over the next two years.

South Fork Snake River Monitoring

Overall, trout populations at both Lorenzo and Conant are doing well; however, species composition is not ideal. In 2009, the rainbow trout abundance was significantly higher than native Yellowstone cutthroat trout at Conant for the first time since monitoring began. Spring flows have been linked to trout abundances in the South Fork Snake River with spring "freshets" that mimic a natural hydrograph having a negative effect on rainbow trout while positively affecting cutthroat trout abundances (Moller and Van Kirk 2003). Moller and Van Kirk (2003) recommend a spring freshet with a maximum freshet flow to minimum winter flow ratio □ 15:1.The maximum to minimum flow ratio for the spring freshet of 2008 exceeded this target with a ratio of 22.6:1. It is possible that not only the magnitude, but also the timing of the spring freshet impacts effectiveness. Since 2004, lower effectiveness of spring freshets impacting rainbow trout recruitment have been observed when the peak of the spring freshet occurred later than June 1 (Figure 5).

Yellowstone cutthroat trout in the South Fork Snake River continue to face obstacles to long-term persistence. The three-pronged management approach aimed at assisting this conservation-reliant species can be effective (Van Kirk et al. 2010). We have observed benefits of spring freshets in 2004 and 2007 when rainbow trout abundance and recruitment was lower than models predicted if no management actions would have taken place (Van Kirk et al. 2010). The timing of the spring freshets should be further evaluated to improve effectiveness. Modifications to tributary weirs and traps continue to improve their effectiveness of maintaining strongholds of Yellowstone cutthroat trout. Main stem harvest of rainbow trout has increased from levels prior to 2004. However, there is room for improvement. Exploitation of rainbow trout needs to be increased to not only deal with the large year class of rainbow trout entering the population from the 2008 spawning season, but also to augment benefits from properly timed spring freshets of adequate magnitude to enhance the spawning success of YCT while negatively affecting RBT spawning activity and success. With adaptive management, continued monitoring, and increases in rainbow trout harvest, the South Fork Snake River will continue to support a healthy population of Yellowstone cutthroat trout.

Literature Cited

Clancy, C. G. 1988. Effects of dewatering on spawning by Yellowstone cutthroat trout in tributaries to the Yellowstone River, Montana. Pages 37-43 *in* R. E. Gresswell, editor. Status and management of interior stocks of cutthroat trout. American Fisheries Society, Symposium 4, Bethesda, Maryland.

Garren, D, W.C. Schrader, D. Keen, and J. Fredericks. 2006. Regional fisheries management investigations. 2004 Job Performance Report, Project F-71-R-29, IDFG 05-15. Idaho Department of Fish and Game. Boise.

Griffith, J. S. 1993. Coldwater streams. Pages 405-425 *in* C. C. Kohler and W. A. Hubert, editors. Inland fisheries management in North America. American Fisheries Society, Bethesda, Maryland.

Henderson, R., J.L. Kershner, and C.A. Toline. 2000. Timing and location of spawning by nonnative wild rainbow trout and native cutthroat trout in the South Fork Snake River, Idaho, with implications for hybridization. North American Journal of Fisheries Management 20:584-596.

Idaho Department of Fish and Game. 2007. Fisheries Management Plan 2007 – 2012. Boise.

Krueger, C. C., and B. May. 1991. Ecological and genetic effects of salmonid introductions in North America. Canadian Journal of Fisheries and Aquatic Sciences 48 (supplement):66-77.

Leary, R.F., F.W. Allendorf, and G.K. Sage. 1995. Hybridization and introgression between introduced and native fish. Pages 91-101 in H.L. Schramm, Jr., and R.G. Piper, editors. Uses and effects of cultured fishes in aquatic ecosystems. American Fisheries Society, Symposium 15, Bethesda, Maryland.

Loomis, J. 2005. The economic values of recreational fishing and boating to visitors and communities along the upper Snake River. Final Report to Trout Unlimited and Henrys Fork Foundation, Colorado State University, Fort Collins.

Meyer, K.A., D.J. Schill, J.A. Lamansky, M.R. Campbell, and C.M. Cegelski. 2006. Status of Yellowstone cutthroat trout in Idaho. Transactions of the American Fisheries Society 135:1329-1347.

Meyer, K. A., J. A. Lamansky, and F. S. Elle. 2008. Lakes and reservoir research. Annual Job Performance Report #08-12. F-73-R-19. Idaho Department of Fish and Game, Boise.

Moller, S., and R. Van Kirk. 2003. Hydrologic alteration and its effect on trout recruitment in the South Fork Snake River. Project Completion Report for Idaho Department of Fish and Game, Idaho State University, Pocatello.

Montana Department of Fish, Wildlife, and Parks. 1997. Mark recapture for Windows (MR5, version 5.0). Montana Department of Fish, Wildlife, and Parks, Helena.

Moore, V., and D. Schill. 1984. South Fork Snake River fisheries investigations. Job Completion Report, Project F-73-R-5. Idaho Department of Fish and Game. Boise.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Fisheries Research Board of Canada, Bulletin 191, Ottawa.

Schill, D.J. 1992. Statewide data summary. Wild trout investigations. Job Performance Report, Project F-73-R-13. Idaho Department of Fish and Game. Boise.

Schrader, W.C., and J. Fredericks. 2006a. South Fork Snake River investigations. 2003 Annual Job Performance Report 06-20. Project F-71-R-28. Idaho Department of Fish and Game. Boise.

Schrader, W.C., and J. Fredericks. 2006b. South Fork Snake River investigations. 2005 Annual Job Performance Report 06-50. Project F-71-R-28. Idaho Department of Fish and Game. Boise.

Schrader, W.C., M. Gamblin, T.J. Herron, and B.A. Rich. 2002. Regional fisheries management investigations. 1995 Job Performance Report, Project F-71-R-20. Idaho Department of Fish and Game. Boise.

Thurow, R.F., C.E. Corsi, and V.K. Moore. 1988. Status, ecology and management of Yellowstone cutthroat trout in the upper Snake River drainage, Idaho. Pages 25-36 in R.E. Greswell, editor. Status and management of interior stocks of cutthroat trout. American Fisheries Society, Symposium 4, Bethesda, Maryland.

USFWS 2006. 12 month status review results. http://edocket.access.gpo.gov/2006/pdf/06-1539.pdf

Van Kirk, R.W., and L. Benjamin. 2001. Status and conservation of salmonids in relation to hydrologic integrity in the Greater Yellowstone Ecosystem. Western North American Naturalist 61:359-374.

Van Kirk, R. W, L. Battle, and W. C. Schrader. 2010. Modelling competition and hybridization between native cutthroat trout and nonnative rainbow and hybrid trout. Journal of Biological Dynamics 4:158-175

Session 6: Resource Extraction and Wild Trout Restoration Efforts

PROPOSED COPPER-SULFIDE MINING IN BRISTOL BAY: IDENTIFIED RISKS TO FISHERIES

Carol Ann Woody

Director, Fisheries Research and Consulting, 6601 Chevigny Street, Anchorage, Alaska, 99502; (907) 248-4776, carolw@alaskalife.net; **Sarah O'Neal**, Associate Director, Fisheries Research and Consulting, 6601 Chevigny Street, Anchorage, Alaska 99502; (907) 248-4776, sarahlouiseoneal@ak.net

ABSTRACT — Bristol Bay is home to one of the largest, most valuable salmon fisheries in the U.S., owing largely to its pristine nature. However, industrial mining interests have claimed over 2,054 km2 in these habitats, threatening two of Bristol Bay's largest salmon producing watersheds, the Kvichak and Nushagak. If the area proposed for mining is developed, the preponderance of peer-reviewed evidence indicates significant risks to fisheries due to habitat degradation and loss.

In Alaska, fish must be explicitly documented in a water body for fish conservation statutes and regulations to apply, though less than one-half of the state's streams have been surveyed. Consequently, presence/absence surveys were conducted in and near mining claims during 2008 and 2009. Data indicate salmon presence in 3 of every 4 headwater streams surveyed. Non-salmon species important to subsistence and sportfishing stakeholders, including Dolly Varden, *Savelinus malma* and rainbow trout *Oncorhynchus mykiss*, were found in 93% of streams surveyed. This study underscores both the importance of headwater streams as essential salmon rearing habitat and the lack of data for two of the world's most productive salmon ecosystems. The work provides some legal protection to 149 km of newly documented salmon streams.

INTRODUCTION

In sharp contrast to salmon declines elsewhere, over 40 million wild salmon returned to Bristol Bay, Alaska in 2009. Bristol Bay is one of the most valuable commercial fisheries in the U.S. (Burgner 1991; NOAA 2010) and is one of the few certified as sustainable (MSC 2009). During 1950 to 2008, U.S. commercial sockeye salmon *Oncorhynchus nerka* landings were valued at about US$7.9 billion dollars with about one-half that value attributed to Bristol Bay stocks (NOAA 2010); sockeye salmon harvests alone have averaged about 30 million annually since 1987 (Sands *et al.* 2008). Bristol Bay also supports a thriving sport fish industry attracting thousands of fishers who generally spend over 90,000 angler days and millions of dollars to catch wild salmon, trout and char from pristine Bristol Bay rivers (Figure 1; Duffield *et al.* 2007; Dye *et al.* 2008).

Noncommercial fishing figures prominently in Bristol Bay communities as well. Athabaskan, Aleut, and Yup'ik peoples annually harvest over 100,000 salmon which they dry, smoke, pickle, salt, can, and store for winter sustenance, as they have for thousands of years. Sockeye salmon are their most important food resource and comprise 60% to 80% of annual subsistence harvests (Fall *et al.* 1996; Fall *et al.* 2006; ADFG 2008a). Non-salmon fish, such as Dolly Varden, rainbow trout *Oncorhynchus mykiss*, and whitefish *Prosopium spp.*, also comprise a significant part of people's diets.

Bristol Bay, Alaska is recognized as one of the world's few remaining Pacific salmon strongholds (Quammen 2009), because wild salmon remain abundant, highly diverse, and their genetic integrity and essential habitats remain intact. Each watershed can contain dozens to hundreds of distinct spawning populations that differ from each other in behavior, appearance, and genetic make up (Hilborn 2003; Ramstad *et al.* 2004; Habicht *et al.* 2007). This high biodiversity, known as the "portfolio effect" (Hilborn *et al.* 2003; Schindler *et al.* 2010), helps ameliorate adverse effects of environmental stressors on salmon production and is considered a major reason Bristol Bay salmon production has remained

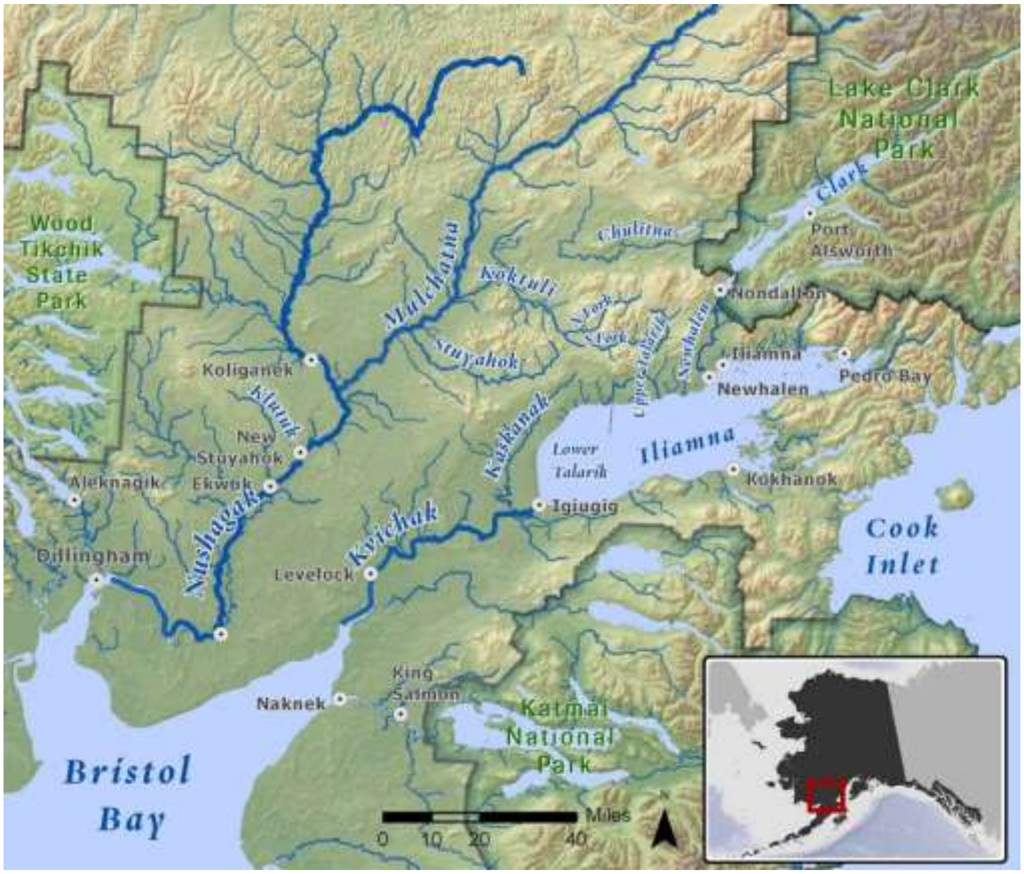


Figure 1. Watersheds and communities (white dots) of Bristol Bay, Alaska.

stable over time, despite changing environmental stressors and heavy exploitation (Hilborn *et al.* 2003).

Future sustainability of Bristol Bay salmon and subsistence fisheries depends, in part, on documenting and conserving essential fish habitats including spawning, incubation, rearing, feeding, and migration habitats. Documentation in Alaska's Anadromous Waters Catalog (AWC) of anadromous fish and their life stage (spawning, rearing, migrating, etc.) is required for certain state permitting requirements to apply. For example, construction of stream crossings, such as culverts or bridges, can be temporally constrained to minimize adverse effects of increased sediment on spawning or incubating fish.

However, basic fish distribution data are lacking for headwater streams draining into the Nushagak and Kvichak rivers, a region slated for large scale copper-sulfide mining (10.8 billion tons, NDM 2010). Mining of this nature has negative impacts on water quality (Kuipers and Maest 2006). Consequently, surveys were conducted in and near the area proposed for mineral development to improve coverage of the State of Alaska AWC in order to afford some protection to fish and their habitat.

Methods

Site Selection

The study focused primarily on first and second order wadeable streams with gradients less than 10%. Selected headwater streams were not listed as anadromous in Alaska State databases, and were generally located in or near mining claims and along proposed road corridors in the Nushagak and Kvichak River drainages. Geographic Positioning System (GPS) coordinates for survey sites were determined using geospatial data layers from the National Hydrography Dataset and the National Elevation Dataset. Because hydrography data sets are sometimes inaccurate (e.g., mapped streams may not exist), final stream selections were determined in the field during low-level helicopter reconnaissance or foot survey. Aerial surveys for adult and spawning salmon were conducted along main stem tributaries (non-wadeable) in mining claims and along proposed road lines where contemporary survey information was lacking.

Fish Surveys

Headwater stream surveys were conducted during the latter half of August and the first week of September using single-pass backpack electrofishing. Fish sampling began by measuring water conductivity, setting appropriate electrofisher parameters, then moving downstream measuring either 150 m or 40 times the stream width, whichever was greater. Survey crews electrofished upstream. If salmon were captured at the study site, the crew electrofished at the most upstream accessible fishable site within that tributary to document uppermost distribution of salmon if possible. Minnow traps were deployed in areas not amenable to electrofishing.

Aerial surveys to substantiate presence of adult or spawning salmon were conducted from a helicopter during September and October in the study area from 30 to 60 m above the stream; and when adult salmon were observed, GPS coordinates were marked and a voucher photo taken.

Habitat Measurements

Habitat measures were based on McCormick and Hughes (1998) and Kaufmann and Robison (1998). One transect was established in a run within each surveyed tributary; GPS coordinates were recorded. Basic water quality was measured in the thalweg with a YSI 556 Multi Probe System (YSI Incorporated, Yellow Springs, Ohio, USA) for temperature, pH, conductivity, and dissolved oxygen (DO). Turbidity was measured using a Hach 2100P Portable Turbidimeter (Hach Company, Loveland, Colorado, USA) and air temperature was measured using a standard alcohol thermometer. Meters were pre- and post-calibrated daily according to manufacturers' instructions. Data produced from meters which failed to meet post-calibration data quality objectives were qualified (Appendix I) and excluded from analyses.

Discharge (cfs) was measured following USGS protocols (Rantz 1982) using Marsh-McBirney Flo-Mate Model 2000 portable meters (Hach Company, Loveland, Colorado, USA). Stream stage was categorized. Morphometric measures included channel width and thalweg measured at both wetted and ordinary high water (OHW) (Kaufman *et al*. 1999). Channel slope was measured using a handheld clinometer and a graduated pole. Visual categorizations were made for both water color (clear, ferric, glacial, humic, muddy, or, in one case yellow) and substrate composition (mm diameter). Upstream and downstream photographs were taken at each transect as well as from ~50 m in the air.

Results

Fish Surveys

Fish were captured in 90 of 92 streams (Figure 2). Three salmon species of two life stages (rearing coho *Oncorhynchus kisutch* and Chinook salmon *O. tshawytscha*, and adult coho and sockeye salmon) were documented at 74% of sites surveyed (excluding sites in the Chulitna River drainage which does not support salmon). Further, 93% (86) of sites surveyed supported subsistence fish (rainbow trout, Dolly Varden, Arctic grayling *Thymallus arcticus*, round whitefish *Prosopium cylindraceum*, burbot *Lota lota*, northern pike *Esox lucius*, and a smelt Osmeridae) or other species (lamprey, slimy sculpin *Cottus cognatus*, ninespine stickleback *Pungitius pungitius* and threespine stickleback *Gasterosteus aculeatus*). No fish were captured at 2% (2) sites. A total of 149 km of essential salmon rearing habitat was nominated for the first time to the State's AWC. The State accepted all nominations. In addition to salmon, one previously undocumented anadromous species was found: an anadromous lamprey *Lampetra camtschatica* and a potentially anadromous smelt (identification pending).

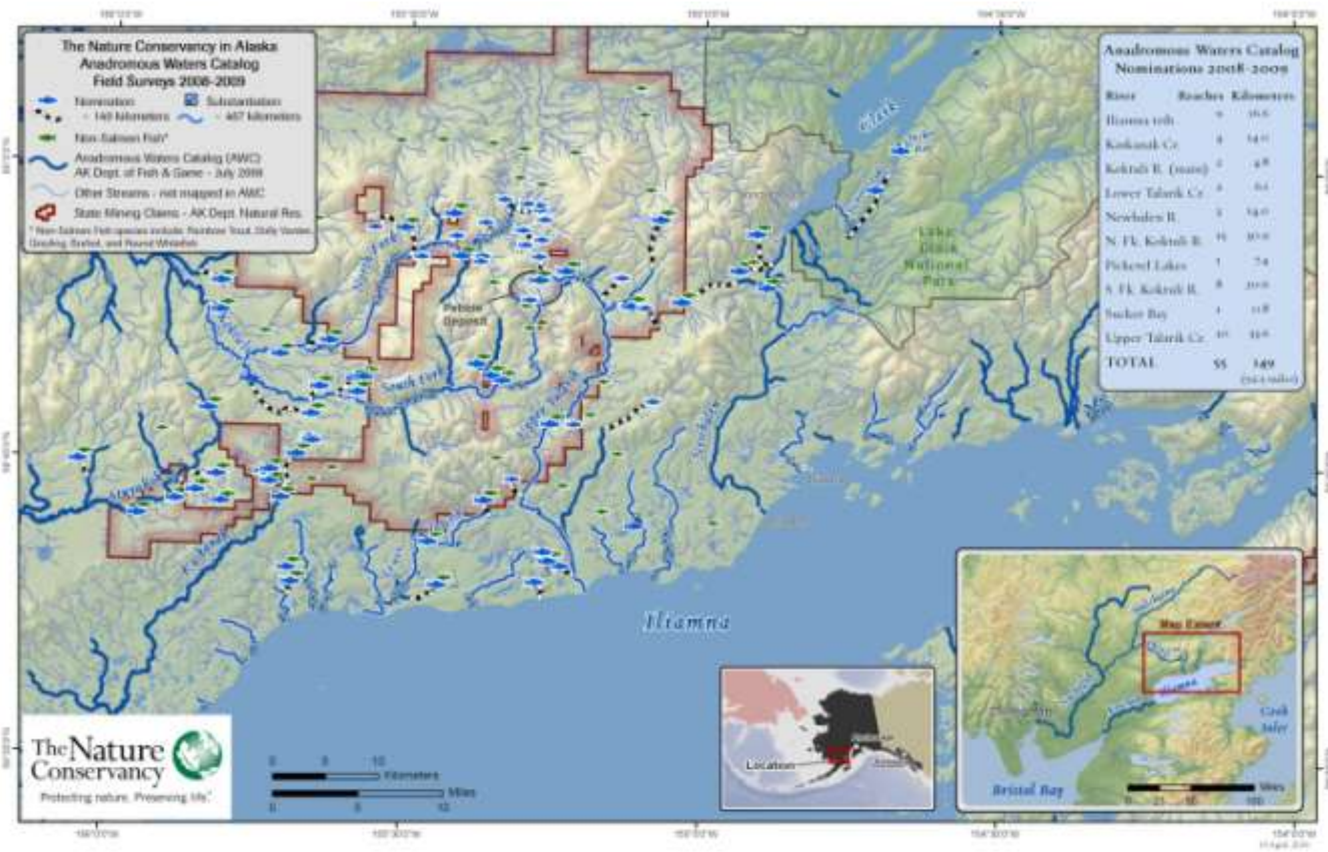


Figure 2. Fish survey results for the Nushagak and Kvichak river watersheds, Bristol Bay, Alaska, 2008 and 2009. Adult salmon presence, indicated as substantiated streams (check marks) was verified during spawning season by helicopter.

During 2008, coho salmon were documented at a channel within the North Fork Koktuli watershed that had no apparent downstream surface connection to a stream. During 2009, several similar sites within the South Fork Koktuli and Stuyahok watersheds also contained coho salmon. Disconnection of those sites may be due to low water in August and resulting down-welling of surface water into the water table, as well as channel constriction creating tunnels covered by dense Sphagnum moss.

Aerial surveys in September and October of 2009, to substantiate presence of adult or spawning salmon in the North and South Fork Koktuli rivers, Upper Talarik Creek, tributaries draining to Lake Iliamna, and along proposed road corridors, resulted in documentation and verification of an additional 408 km of anadromous rivers (Figure 2). Some evidence of spawning was documented during those surveys as well.

Habitat Measurements

Surveyed headwater tributaries in both years were generally first- or second-order streams, with cold, clear water (temperatures averaged 7.7 °C in 2008 and 8.8 °C in 2009), nearly neutral pH (averaging 7.30 in 2008 and 7.09 in 2009) and low conductivity (averaging 58 µS/cm in 2008 and 44 µS/cm in 2009). Dissolved oxygen levels were at or near saturation (averaging 11.1 mg/L in 2008 and 11.4 mg/L in 2009).

In 2008, wetted stream widths averaged 1.9 m wide by 25.7 cm deep compared to ordinary high water (OHW) which averaged 2.2 m wide by 35.8 cm deep measured at the thalweg. Discharge averaged 0.04m^3/s cfs at the low to medium flows encountered in 2008. Fourteen of 23 streams had ≥ 50% substrates comprised of fine to coarse gravel (≥2 mm dia to < 64 mm dia). In 2009, wetted widths averaged 4.1 m wide by 30 cm deep; ordinary high water (OHW) averaged 5.0 m wide by 50 cm deep measured at the thalweg. Discharge averaged 0.33 m^3/s (Table 4). Most streams had ≥ 50% substrates comprised of fine to coarse gravel (≥ 2 mm dia to < 64 mm dia).

Discussion

This study documented over 149 km of essential salmon rearing habitats which were subsequently nominated for the first time to the State of Alaska Anadromous Waters Catalog (AWC). All nominations were accepted by the State for inclusion in the AWC. Combined results from both years showed 93% of all surveyed streams contained fish, and anadromous salmon presence in 74% of headwater tributaries, excluding the Chulitna River. Our surveys within the Chulitna River watershed did not reveal salmon presence, possibly due to habitat factors including slow-moving water, fine sediments, and presence of large, abundant, predacious northern pike. A total of thirteen species were documented during the study.

In addition to juvenile surveys, aerial surveys for spawning adults resulted in documentation and verification of another 408 km of anadromous rivers. Although it was not always possible to verify spawning, evidence of redds was apparent in drainages surveyed. However, other studies conducted in Lake Clark, in the Kvichak drainage, indicate the presence of adult sockeye salmon in a tributary signify spawning will occur in that tributary (Young and Woody 2007). A report by an environmental consulting company (HDR 2008) also documented the presence of juvenile salmon in the upper reaches of the North and South Fork Koktuli rivers and Upper Talarik Creek during winter surveys, suggesting that spawning occurs in these systems. Additional adult surveys to verify spawning and winter studies focused on juvenile presence in these systems would provide additional insight.

These surveys were conducted during periods of summer low flows. Thus, fishless survey sites may support fish at periods of higher flow. As evidenced by the presence of salmon in sites disconnected from surface water flow in this study, groundwater supplied refuges during periods of low flow. Annual floods during spring and fall in Bristol Bay, likely reconnect such refugia allowing salmon to move among ephemeral habitats. Limited field observations indicate fish are able to exploit hyporheic corridors for extended distances due to the large alluvial substrates and copious groundwater in the study region (Boulton *et al.* 1998). The lack of additional information on salmon use of ephemeral habitats in the region and the extent to which groundwater provides refuge warrants further investigation. The functional significance, flow patterns, and vulnerability of the hyporheic zone to proposed mining would provide insight into potential impacts groundwater contamination could have on salmon production.

Coho salmon were the most commonly observed salmon during the study. The paucity of Chinook salmon observations is likely due to habitat segregation as Chinook salmon prefer larger, main stem habitats (Lister and Genoe 1970; Murphy 1989; Scarnecchia and Roper 2000). Because our study was generally limited to small, shallow, headwater tributaries during low flow, Chinook salmon may be more abundant in deeper, higher velocity habitats not surveyed for this study.

Non-salmon fish species are an important subsistence food resource for people in this region. Subsistence consumption- surveys for the Kvichak River watershed showed increased harvest of Dolly Varden for subsistence over the last decade (Krieg *et al.* 2005). The life history of Dolly Varden in this region is poorly understood, including their anadromous tendencies, movement patterns, and abundance. However, preliminary results of otolith microchemistry analysis for the study area indicate low levels (2 of 29 samples in the North and South Fork Koktuli rivers) of Dolly Varden anadromy (Christian Zimmerman, personal communication,). Further, radiotelemetry studies in a nearby watershed proved anadromy to be common in Dolly Varden (Lisac and Nelle 2000). Because Dolly Varden was the second most abundant species encountered in this study (after sculpin), and because it is such an important subsistence species, further information on their life history patterns in the Nushagak and Kvichak River watersheds would provide valuable information for their conservation.

Small headwater streams are often assumed not to be important salmon producing habitats in Alaska, although collectively they produce millions of salmon and determine water flow and chemistry of larger rivers. As illustrated by this and numerous other studies, headwaters comprise a significant proportion of essential spawning and rearing habitat for salmon and non-salmon species, all of which are important to subsistence users in the region. Because mortality is highest during a fish's early life , successful negotiation of the vulnerable juvenile life history phase increases the probability an individual will survive to reproduce—a key factor in a sustainable fishery.

In addition to documenting fish presence and absence, this study evaluated basic water quality and habitat parameters, indicating pristine conditions throughout the area. Temperatures are cool, well below upper tolerance limits for all species and life stages of salmon (Richter and Kolmes 2005). Oxygen levels were at or near saturation at all sites, and well above critical levels for egg incubation and juvenile rearing as well as spawning (Quinn 2005). Likewise conductivity is low in the region, closer to that of distilled water or melted snow than typical freshwater levels (CWT 2004). Finally, pH values were near neutral. Lower pH increases the solubility of toxic metals (Dodds 2002), and is a common impact associated with copper-sulfide mining. In addition to chemical habitat parameters, physical habitat parameters suggested abundant spawning gravel, and depths appropriate for both spawning and rearing (Quinn 2005).

It is important to note that all fish and habitat measurements represent only two brief snapshots in time. A more robust dataset including diel temperature, oxygen, conductivity patterns, as well as intra-annual fish distribution throughout the year would be useful in understanding ranges of those parameters.

Conservation of the world's largest, most valuable, sustainable, wild salmon fisheries in Bristol Bay depends, in part, on conserving the diverse habitats essential to fish survival and reproduction. The information presented here on fish distribution, headwater stream chemistry, and channel morphology provides more complete and accurate information for future fish conservation decisions. However, thousands of similar streams that contribute to fish production remain un-surveyed, and therefore are not afforded statutory protection offered by inclusion in the AWC in regions proposed for mineral development.

Acknowledgements

Funding and logistical support were provided by The Nature Conservancy; The Gordon and Betty Moore Foundation; Trout Unlimited; The Southwest Alaska Salmon Habitat Partnership with the U.S. Fish and Wildlife Service; The National Fish and Wildlife Foundation with the Bureau of Land Management; the Wallace Research Foundation, Mr. and Mrs. John Lovelace, Bristol Bay Regional Seafood Development Association and the National Parks Conservation Association. Justin Arnold, Steve Baird, Daniel Chythlook, Jason Gray, Stan Hermens, Kristy Jeffries, J. Johnson, Sue Mauger, Dan Prince, Joel Reynolds Dan Rinella, Jane Whitsett, Dan Young, and June and Bob Tracey provided invaluable assistance both in and out of the field. Marcus Geist produced the Geographic Information System (GIS)-based maps and data layers for nominations to Alaska's Anadromous Waters Catalog; Laura Sorensen assisted with quality assurance. A special thanks to the people of Nondalton and the Nondalton Tribal Council.

Literature Cited

Boulton, A. J., S.F. Findlay, P. Marmonier, E. H. Stanley, and H. M. Valett. 1998. The functional significance of the hyporheic zone in streams and rivers. Annual Review of Ecology and Systematics. 29:58-81.

Burgner, R. L. 1991. Life history of sockeye salmon (*Oncorhynchus nerka*). Pacific Salmon Life Histories. C. Groot and L. Margolis (editors). Published by UBC Press, Canada.

CWT (Clean Water Team) 2004. Electrical conductivity/salinity Fact Sheet, FS-3.1.3.0(EC). in: The Clean Water Team Guidance Compendium for Watershed Monitoring and Assessment, Version 2.0. Division of Water Quality, California State Water Resources Control Board (SWRCB), Sacramento.

Dodds, W.K. 2002. Freshwater ecology: Concepts and environmental applications. Academic Press, San Diego, CA.

Duffield, J., D. Patterson and C. Neher. 2007. Economics of wild salmon watersheds: Bristol Bay, Alaska. Available from J. Duffield: Department of Mathematical Sciences, University of Montana, Missoula.

Dye, J. E., C. J. Schwanke, and T. A. Jaecks. 2008. Report to the Alaska Board of Fisheries for recreational fisheries of Bristol Bay, 2004, 2005, and 2006. Alaska Department of Fish and Game. Special publication No. 06-29. Anchorage.

Fall, J. A., M. B. Chythlook, J. C. Schichnes, and J. M. Morris. 1996. An overview of the harvest and use of freshwater fish by the communities of the Bristol Bay region, southwest Alaska. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper Series. Technical Paper No. 166, 171 p.

Fall, J.A., D.L. Holen, B. Davis, T. Krieg, and D. Koster. 2006. Subsistence harvests and uses of wild resources in Iliamna, Newhalen, Nondalton, Pedro Bay, and Port Alsworth, Alaska, 2004. Alaska Department of Fish and Game, Division of Subsistence Technical Paper No. 302. Juneau.

Habicht, C., L.W. Seeb, and J.E. Seeb. 2007. Genetic and ecological divergence defines population structure of sockeye salmon populations returning to Bristol Bay, Alaska, and provides a tool for admixture analysis. *Transactions of the American Fisheries Society* 136: 82-94

HDR. December 2008. Completion report for fisheries studies conducted under fish resource permits SF-2004-061, SF-2004-114, SF-2005-049, SF-2006-057, SF-2007-053, AND SF-2007-144 as amended. For the Pebble Project. 42p. Available from the Alaska Department of Fish and Game, Anchorage**.**

Hilborn, R., T. P. Quinn, D. E. Schindler, and D. E. Rogers. 2003. Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences of the United States of America 100(11):6564-6568.

Kaufmann, P.R. and E.G. Robison. 1998. Physical Habitat Characterization. pp 77-118 In: J.M. Lazorchak, D.J. Klemm and D.V. Peck (editors.). Environmental Monitoring and Assessment Program -- Surface Waters: Field Operations and Methods for Measuring the Ecological Condition of Wadeable Streams. EPA/620/R-94/004F. U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C.

Kaufmann, P. R., P. Levine, E. G. Robison, C. Seeliger, and D. V. Peck. 1999. Quantifying physical habitat in wadeable streams. Environmental Protection Agency, Report 620/R-99/003, Corvallis, Oregon.

Krieg, T., M. Chythlook, P. Coiley-Kenner, D. Holen, K. Kamletz, and H. Nicholson. 2005. Subsistence Fisheries Assessment: Kvichak River Watershed Resident Species. Federal Subsistence Fishery Monitoring Program, Final Project Report No. FIS 02-034. U. S. Fish and Wildlife Service, Office of Subsistence Management, Fisheries Resource Monitoring Program, Fishery Information Service, Anchorage, Alaska.

Kuipers, J.R., Maest, A.S., MacHardy, K.A., and Lawson, G. 2006. Comparison of Predicted and Actual WaterQuality at Hardrock Mines: The reliability of predictions in Environmental Impact Statements. Earthworks. Washington, D.C.

Lisac, M. J. and R. D. Nelle. 2000. Migratory behavior and seasonal distribution of Dolly Varden *Salvelinus malma* in the Togiak River watershed, Togiak National Wildlife Refuge. Final Report. U.S. Fish and Wildlife Service. Dillingham, Alaska.Lister, D. B. and H. S. Genoe. 1970. Stream habitat utilization by cohabitating underyearling Chinook and coho salmonids. Journal of the Fisheries Research Board of Canada. 27:1215-1224.

MSC (Marine Stewardship Council). 2009. Alaska salmon sustainable certification available at: http://www.msc.org/track-a-fishery

McCormick, F. H., and R. M. Hughes. 1998. Aquatic vertebrates. Pages 161 – 182 in J. M. Lazorchak, D. J. Klemm, and D.V. Peck, editors. 1998. Environmental monitoring and assessment program surface waters: field operations and methods for measuring the ecological condition of wadeable streams. EPA/620/R-94/004F. U.S. Environmental Protection Agency, Washington, D.C.

Murphy, M.L., J. Heifetz, J.F. Thedings, S.W. Johnson, and K.V. Koski. 1989. Habitat utilization by juvenile Pacific salmon (*Onchorynchus*) in the glacial Taku River, southeast Alaska. Canadian Journal of Fisheries and Aquatic Sciences 46: 1677-1685.

NDM (Northern Dynasty Minerals). 2010. The Pebble Project: The future of U.S. Mining & Minerals Fact Sheet. http://www.northerndynastyminerals.com/ndm/Home.asp. Accessed August 3, 2010.

NOAA. March 2010. Annual commercial landings by group. Queried database, available at http://www.st.nmfs.noaa.gov/st1/commercial/landings/gc_runc.html accessed 12 March 2010.

Quammen, D. Photographs by R. Olson. August 2008. Where the Salmon Rule. National Geographic 216: 29 – 55.

Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. American Fisheries Society, Bethesda, MD.

Ramstad, K. M., C. A. Woody, G. K. Sage, and F. W. Allendorf. 2004. Founding events influence genetic population structure of sockeye salmon (*Oncorhynchus nerka*) in Lake Clark, Alaska. Molecular Ecology 13:277-290.

Rantz, S. E. *et al*. 1982. Measurement and computation of stream flow: Volume II, computation of discharge. U.S. Geological Survey Water Supply Paper 2175: 285-631.

Richter, A. and S.A. Kolmes. 2005. Maximum temperature limits for Chinook, coho, and chum salmon, and steelhead trout in the Pacific Northwest. Reviews in Fisheries Science 13: 23-49.

Sands, T., C. Westing, P. Salomone, S. Morstad, T. Baker, F. West, and C. Brazil. 2008. 2007 Bristol Bay area annual management report. Alaska Department of Fish and Game, Fishery Management Report No. 08-28, Anchorage.

Scarnecchia, D. L., and B. B. Roper. 2000. Large-scale, differential summer habitat use of three anadromous salmonids in a large river basin in Oregon, USA. Fisheries Management and Ecology 7(3):197-209.

Schindler, D.E., R. Hilborn, B. Chasco, C.P. Boatright, T.P. Quinn, L.A. Rogers, and M.S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465: 609–612

Young, D. B. and C. A. Woody. 2007. Dynamic in-lake spawning migrations by female sockeye salmon. Ecology of Freshwater Fish. 16: 155–164.

.

Spatial Analysis of Land-use Activity on Westslope Cutthroat Trout: Implications for Policy and Management

E.J. Valdal

Team Lead, Integrated Land Management Bureau, Province of British Columbia, 441 Columbia St. Kamloops, BC, CANADA, V2C 2T3. Ph: (250) 828-4354, eric.valdal@gov.bc.ca

M.S. Quinn

Associate Professor, Faculty of Environmental Design, University of Calgary, 2500 University Drive N.W., Calgary, AB, CANADA, T2N 1N4. Ph: (403)220-7013, FAX: (403)284-4399, quinn@ucalgary.ca

ABSTRACT — A landscape scale analysis of anthropogenic and natural disturbance indicates a significant negative relationship between the cumulative effects of forestry related activities and the relative abundance of westslope cutthroat trout *Oncorhynchus clarkii lewisi* in southeastern British Columbia. Cutthroat trout abundance within six Kootenay River headwater streams ranged from 0.00 to 0.0325 fish/m^2. Univariate analyses within a blocked regression design resulted in statistically significant ($p<0.05$) negative relationships between cutthroat trout abundance and road density, roads on erodible soils, roads within near-stream zones, and two measures of logging proximity to the stream edge. There was no statistically significant relationship between cutthroat trout abundance and total area logged (km^2/km^2). Roads over erodible soils within near-stream zones emerged as the most significant individual variable. A multivariate model to predict cutthroat trout abundance included roads within near-stream zones and recent logging adjacent to streams. Evidence from this study indicates that logging of non-fish bearing perennial and ephemeral streams is a key factor that has negative downstream effects on cutthroat trout abundance. Contrary to many conventional forest management approaches, this study suggests that considering the spatial distribution of disturbance should take precedence over the total amount of disturbance and that disturbance types can accumulate to produce negative effects on cutthroat trout abundance.

Introduction

Research associating landscape scale condition and westslope cutthroat trout *Oncorhynchus clarkii lewisi* is lacking to support the effective conservation of this cutthroat subspecies. Stream fish in particular are at risk from dispersed land use activity over broad landscapes, because watersheds accumulate and concentrate the effects of disturbances over space and through time (Fausch et al. 2002). Historically, site scale habitat parameters were thought to regulate fish populations in lotic systems (Creque et al. 2005), and the site scale approach has dominated stream fish research, industrial development approvals and impact assessments. However, this focus does not adequately account for the effects of land use within the overall watershed, thus limiting effective conservation (Schlosser and Angermeier 1995). The purpose of this research was to conduct a broad scale spatial analysis of landscape disturbance and relative westslope cutthroat trout abundance.

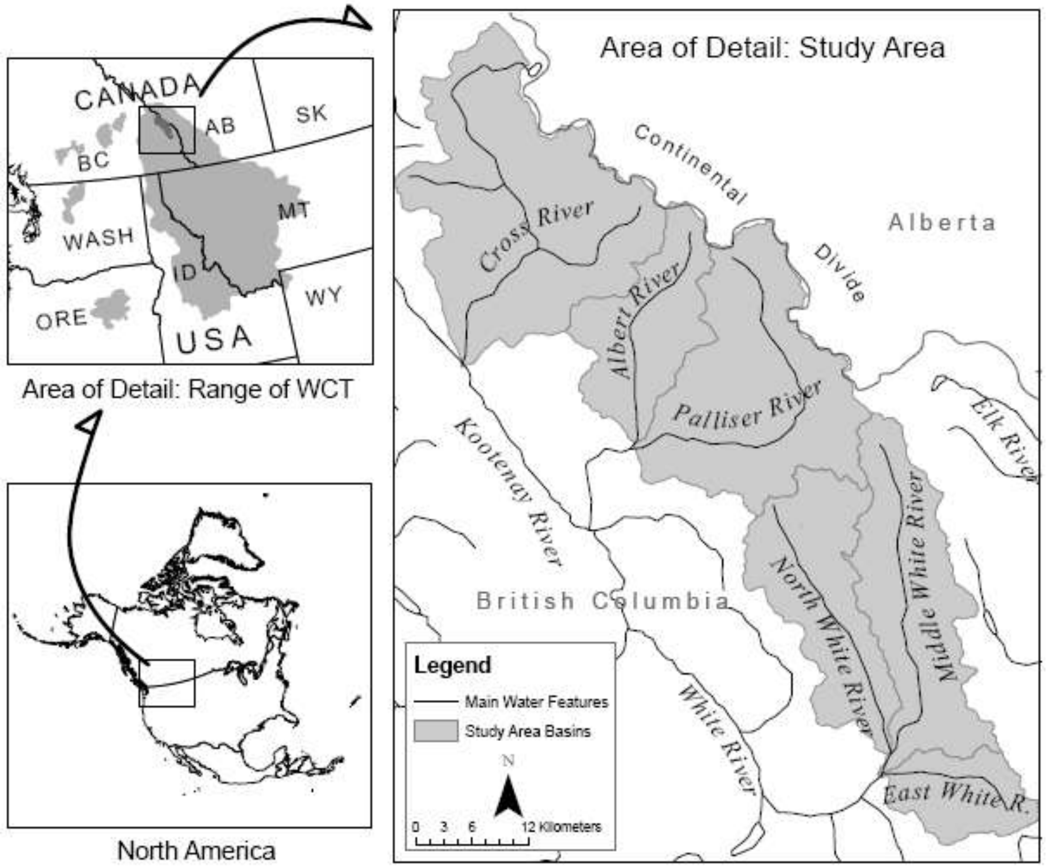


Figure 1: Range of WCT and Project Study Area

Methods

Study Area

The study area for this project was the upper portion of the Kootenay River Watershed in British Columbia (Figure 1). The area is in the northern periphery of westslope cutthroat trout range; however,

Costello and Rubidge (2003) consider the British Columbia range portion to be a stronghold of the subspecies. The study area encompasses 185,700 ha over six river basins that drain into the Kootenay River, a major tributary of the Columbia River. It consists of four 5th order watersheds and two 6th order basins that include the Albert, Cross, East White, Middle White, North White, and Palliser rivers. The Albert, Cross, and Palliser rivers have natural stream barriers at their downstream ends. Elevations range from 1100 to1300 m on the valley floors to 3400 - 3600 m at the mountain peaks. Since the early 1960s, logging has been the dominant land use within the study area and wildfire has been the most extensive natural disturbance.

Data Sources, Quality and Selection

Cutthroat Trout Abundance

We used existing landscape and fish and fish habitat inventories collected by the British Columbia (BC) government agencies to examine associations between cutthroat trout abundance and landscape variables. The BC Interior Watershed Analysis Procedure (IWAP, (BCMoF 1995b)) was used as a framework for analysis. Reconnaissance (1:20,000) Fish and Fish Habitat Inventory (FFHIP) sample datasets were acquired from the BC Ministry of Environment (MOE) for six 5th and 6th order basins within the Upper Kootenay Watershed. The single-pass electrofishing surveys followed a stratified random sample design covering whole watersheds, with stream reach and basins defined from 1:20,000 scale maps and aerial photos (BCFisheries 2001). Within the six watersheds of the study area, sub-basins and sub-watershed zones were distinguished as units of analysis.

The data included 317 samples covering more than 27 km of stream length. Samples for basins were collected between 1996 and 2000. Age-0 fish (length < 70 mm) were removed from the samples to minimize seasonal differences amongst streams and account for unequal sampling of fry between sub-watersheds. Remaining fish (n=312) were grouped into three size classes: 70-149 mm, 150-219 mm and 220+ mm. Single pass electrofishing potentially introduces bias when comparing relative abundance of cutthroat trout across streams of different size; therefore, we applied a sample correction factor for our study area based on input from provincial fish inventory experts (Valdal 2006) and independently tested this with a correction factor from a similar WCT abundance study in another area (Dunnigan et al. 1998).

Cutthroat trout relative abundance values (WCT/m²) were generated for each sub-basin and sub-watershed zone in stream reaches that were <=10% average gradient (Valdal and Quinn 2010).

Catchment Condition

The BC Interior Watershed Analysis Procedure (IWAP) Level One analysis is a GIS based approach that assesses hydrological impacts in a watershed (BCMoF 1995b). Many of the indicators used in the Level One IWAP are the same variables used in other broad-scale salmonid habitat research (e.g., Eaglin and Hubert 1993; Baxter et al. 1999; Bisson et al. 2002). The source data include forest cover, density of logging roads, soils, digital elevation model, hydrology, terrain hazard, and watersheds. The IWAP assesses the potential for certain hydrological impacts in an area of interest, specifically the potential for changes in peak flows (equivalent clear-cut area), accelerated surface erosion (road density variables and stream crossing density), and changes to riparian zones (logging to stream bank of fish bearing and non-fish bearing streams) using landscape disturbance variables. The IWAP variables included the first eight variables listed in Table 1. Four additional variables were added to the IWAP list based on other published salmonid habitat research (last 4 variables in Table 1). Data layers were combined and queried by watershed using ArcGIS (Environmental Systems Research Institute 2002).

Spatial Analysis

Blocked statistical designs have been applied to a variety of ecological investigations where sources of heterogeneity among experimental units need to be accounted for (Schwarz 2005). A blocked design (blocking factor was watershed) was selected for this project to account for hypothesized differences in resident westslope cutthroat trout populations among the six watersheds that comprise the study area. The GLM Univariate method (SPSS 2004) was employed for multiple regression analysis. Relative cutthroat trout abundance was assigned as the dependent variable and a variable indicating parent watershed was used as a fixed factor (blocks). Regressions of single predictors were carried out for all variables.

Results

Cutthroat Trout Abundance

Relative abundance varied among sub-basins and sub-watershed zones from 0.00 to 0.0325 cutthroat trout/m². The highest mean abundances were found in the Albert River watershed (mean = 0.022 cutthroat trout/m²) and the lowest found in the Middle White watershed (mean = 0.0018 /m²). Cutthroat trout abundances in the Albert, East White and North White rivers were similar to abundances found in other research of cutthroat trout headwater habitats (e.g., 0.003 fish/m² to 0.038 fish/m² in the North Fork of the Coeur d' Alene River;.

Analysis of Catchment Condition and Cutthroat Trout Abundance

Road density on erodible soils within 100 m of stream was the most important single variable predictor of relative cutthroat trout abundance (Table 1). Only models with significant results ($p < 0.05$) have the model R^2 displayed. A negative sign in front of the R^2 value indicates a negative relationship.

Table 1. Summary of statistical output for individual predictors of relative cutthroat trout abundance. Model strength (R^2) values are provided for results that are statistically significant ($p < 0.05$).

Variable	Variable Description	F	p	df	R^2
ECA	Equivalent Clear-cut Area (km²\/km²)	0.695	0.414	1,21	
RDDEN	Road Density (km\/km²)	6.917	0.016	1,21	-0.637
RDEROD	Roads on Erodible Soils (km/ km²)	4.406	0.057	1,21	-0.595
RDSTR	Roads within 100 m. of a Stream (km/ km²)	9.515	0.006	1,21	-0.668
RDERDSTR	Roads on Erodible Soils within 100m. of Stream (km/ km²)	14.67	0.001	1,21	-0.716
STCRDEN	Stream Crossing Density (number/km²)	2.678	0.117	1,21	
FBSTRLOG	Portion of Fish Bearing Streams Logged to Stream-bank (km\/km)	3.167	0.091	1,19	
ASTRLOG	Portion of All Streams Logged to Stream-bank (logging since 1960) (km/ km)	8.23	0.01	1,19	-0.656
ASTNEWLOG	Portion of All Streams Recently Logged to Stream-bank (ECA = 0)) (km/ km)	6.818	0.017	1,19	-0.637
ASTDISTRB	Portion of All Streams Disturbed to Stream-bank (Logging & Wildfire) (km\/km)	0.2	0.66	1,19	
AVGH2O	Average Sub-basin Water Temperature (at time of fish sampling) (°C)	0.216	0.157	1,21	
AVGGRAD	Sub-basin Average Fish-bearing Gradient (%)	0.001	0.971	1,19	

Significant relationships were found between six landscape variables and cutthroat trout relative abundance. In order of significance from strongest to weakest these were (1) roads on erodible soils within 100 m of a stream (RDERDSTR), (2) roads within 100 m of a stream (RDSTR), (3) portion of all streams logged to stream-bank (ASTRLOG), (4) overall road density (RDDEN), (5) portion of all streams recently logged to the stream bank (ASTNEWLOG), and (6) roads on erodible soils (RDEROD).

Abundance Model

The best multiple regression model ($R^2 = -0.756$, $n = 25$) included only two variables: roads within 100 m of a stream (RDSTR) and recently logged streams (ASTNEWLOG). Other models were not significant or the variables were correlated and resulted in one variable masking the other.

Discussion

Distributions of stream fishes at a landscape scale may be influenced by such factors as water flow, stream temperature, displacement by non-native trout, and the quality and distribution of critical habitats (Paul and Post 2001; Rieman et al. 2001; Rubidge et al. 2001; Pess et al. 2002; Dunham et al. 2003; Creque et al. 2005). In the current study, detectable differences of flow, temperature and displacement factors did not account for variation in cutthroat trout abundance. Rather, attributes of catchment condition were the significant factors influencing resident cutthroat trout distributions.

Factors Affecting Cutthroat Trout Abundance

Our study indicates that road systems significantly influence assemblages of resident cutthroat trout. Forestry roads are well known to provide the mechanisms for accelerated sediment production and delivery to stream channels within watersheds (Anderson 1998; Carver 2001). Carver (2001) reported increased loads of fine sediment are detrimental to salmonids, causing elevated turbidity and increased stream embeddedness. The latter generally results in loss of spawning habitats and habitat for cutthroat trout food sources, such as aquatic invertebrates. The significant negative relationship between road density and cutthroat trout abundance is consistent with the results of other fish and fish habitat research (Dunnigan et al. 1998; Costello and Rubidge 2003; Baxter et al. 1999).

The highly significant result of road density within 100 m of streams (RDSTR) in relation to cutthroat trout abundance provides evidence that the spatial arrangement of roads plays a large role in affecting cutthroat trout. This variable was more significant than the road density variable Carver (2001) states that stream buffers can prevent elevated loads of sediment from reaching the stream channel; thus, roads should be located outside of the buffer zone to minimize the potential for sediment delivery. Our results support the contention that the potential for negative effects on cutthroat trout abundance is a function of road density close to streams and not simply road density in the catchment (Schiess and Krogstad 2004).

The roads over erodible soils variable (RDEROD) (e.g. silty clay loam; Valdal 2006) is a key variable for understanding distributions of cutthroat trout. The most significant co-predictor of the dependent variable was road density on erodible soils within 100 m of streams (RDERDSTR). This outcome provides the strongest evidence that the linkage between lower levels of cutthroat trout abundance and higher levels of road disturbance is most likely sedimentation. Barrett et al. (1998) found that negative effects from forestry to salmonid habitats were more dependent on surficial geology and soils than other measures of industrial activity. Synthesis of the results from RDEROD, RDSTR and RDERDSTR suggest that both road-stream proximity and erodibility of the soil are extremely important factors to consider to minimize negative forestry-fish interactions.

Correlative evidence from this study suggests that logging activities within near-stream zones potentially result in negative effects on cutthroat trout abundance. Comparisons between statistically significant and non-significant near-stream logging related variables indicate that differences in stream classes, disturbance types and cumulative disturbance duration, play important roles in resident cutthroat trout assemblages.

The portion of fish bearing streams that were logged to the stream bank since 1960 (FBSTRLOG) was not significantly correlated with cutthroat trout abundance. Most fish bearing streams have riparian reserve zones that buffer the stream bank for aquatic habitat protection (BCMoF 1995c). Conversely, the portion of logging on all streams (ASTRLOG) was significant ($p = 0.01$), as was the portion recent logging of all streams (ASTNEWLOG, $p = 0.017$). The key difference between FBSTRLOG and the latter two variables is the inclusion of logging disturbance over perennial and ephemeral non-fish bearing streams. The significance of ASTRLOG and ASTNEWLOG suggests that logging of perennial and ephemeral non-fish bearing streams has significant negative downstream consequences for cutthroat trout.

It is well established within geomorphic literature that land use disturbance in upstream reaches results in impacts to channel morphology and in-stream habitat downstream (Rieman et al. 2001; Kondolf et al. 2002). In a literature synthesis of forestry-riparian interactions, Carver (2001) concluded that it is important to maintain riparian vegetation on ‘small’ headwater streams due to temperature, nutrient and sediment delivery implications downstream.

The non-significance of the portion of logging and wildfire disturbance of all stream classes (ASTDISTRB) is particularly interesting. The amount of disturbance adjacent to streams increased by 40% when wildfire was included over and above logging, yet the increase did not account for variation of cutthroat trout abundance. Other research has found differences in aquatic in-stream habitat and native trout assemblages between landscapes affected by fire (only) and those affected by logging. In northern Idaho, Huntington (1998) found relatively higher habitat quality and native trout abundance in basins with a history of wildfire (i.e., < 100 years) versus basins that were disturbed by forestry operations in the same time period. However, fire severity and salvage logging of burned stands are important factors that influence the degree of aquatic ecosystem damage within a watershed (Bescheta et al. 1995; Dunham et al. 2003; Karr et al. 2004).

Although fire severity data were not available for this project, differences in severity amongst and between fires may result in different effects on fish habitat. Karr et al. (2004) suggested that adaptations by riparian and aquatic ecosystems may assist to moderate the effects of wildfire. Dwire and Kauffman (2003) found that riparian areas frequently differ from adjacent uplands in vegetative composition and structure, geomorphology, hydrology, microclimate, and fuel characteristics that often contribute to lower severity burns. Lower severity burns may allow retention of the physical mechanisms that assist to moderate stream sediment input and changes to the nutrient and temperature regimes (Dwire and Kauffman 2003). This phenomenon may contribute to the lack of effect on cutthroat trout abundance by streams disturbed by wildfire in this study.

The area classified as wildfire did not include those areas that were burned and then salvage logged. Approximately 16% of logged areas were preceded by wildfire within the project area. Salvage logging typically prevents natural recovery processes and may worsen degraded aquatic conditions (Karr et al. 2004). These impacts tend to have a cumulative effect because fire-affected ecosystems are more sensitive to further disturbances and amplify the negative ecological effects of logging (Bescheta et al. 1995). Salvage logging within riparian areas may function to amplify the contrast between logging and wildfire.

The lack of a relationship between equivalent clear-cut area and cutthroat trout abundance is interesting, given that this indicator has been a focal point for the management and monitoring forest-fish interactions throughout BC. The equivalent clear-cut area result is consistent with the salmonid habitat-association research of Dunnigan et al. (1998) in Idaho and Barrett et al. (1998) in Northern California. Barrett et al. (1998) suggested that the impacts to fish habitats were more closely tied to variation in surficial geology and soils than to measures of industrial activity alone.

Cumulative Effects of Disturbance on Cutthroat Trout Abundance

Forestry activity within near-stream zones is well known to elevate the risk of potential negative effects to fish habitats (Eaglin and Hubert 1993; BCMoF 1995b; Anderson 1998; Carver 2001). This notion is strongly supported by the significant multiple regression model for cutthroat trout abundance that included roads within near-stream zones (RDSTR), portion of streams with recent logging (ASTNEWLOG) and the blocked factor as covariates. This model captures the negative cumulative effects from roads and logging on cutthroat trout abundance. Critical to this relationship is the role to which the spatial arrangement of disturbance is a factor. Within the geographic scope of these variables, there was no significant correlation between roads and logging.

Consideration of the single and multiple predictor results suggest that incremental site-scale habitat protection policy has negatively accumulated to become a limiting factor for cutthroat trout. Creasy (1998) suggested that the problem with cumulative effects generally rests within the scope of routine decision-making policy rather than a problem of environmental science. Incremental site-scale habitat protection policy implementation does not totally account for cumulative impacts to cutthroat trout and its habitat. The project results clearly indicate that logging of non-fish bearing perennial and ephemeral streams is a major limiting factor to the conservation of cutthroat trout.

A prevailing belief is that negative effects do not occur within a watershed until its equivalent clear-cut area is greater than 25-30% (Eaglin and Hubert 1993). Since catchment condition has a strong influence on fish populations (Frissell et al. 1986), the results of this project contradict the 30% equivalent clear-cut area watershed assessment threshold in many ways. First, equivalent clear-cut area was not demonstrated to be a useful predictor of habitat quality. Second, the equivalent clear-cut area within this project ranged from zero to 27% (mean = 8%). However, the negative effects of other disturbance variables were clearly detectable using relative abundance of cutthroat trout. In watersheds with high variation of cutthroat trout abundance (i.e., Albert, North and East White rivers), the lowest fish densities occurred concurrent with equivalent clear-cut areas that ranged from 5-17%. The results indicate that the disturbance indicator and threshold (30%) for landscape scale watershed assessments, contribute to the cumulative negative effect of incremental site-scale habitat protection measures by being ineffective.

Application of Landscape Level Habitat Relationships

Central to the development of conservation strategies for cutthroat trout is the quantitative understanding of the relationship between fish abundance and landscape scale habitat condition (Pess et al. 2002; King et al. 2005). However, associations developed at the landscape scale do not directly establish cause and effect relationships. Landscape level disturbance is only indirectly linked to stream biota via “a dizzying array of near-stream and in-stream abiotic factors” (King et al. 2005). That being said, the benefits for linking landscape disturbance to stream condition via specific threats for the express purpose of resource management and conservation planning, far outweighs issues of independence. This is especially valid if the limitations are made transparent and if the habitat-associations are appropriately applied in concert with site level habitat understanding (Creque et al. 2005). The results of this study should compliment current site level fish habitat understanding in applications of future forestry development conservation strategies for cutthroat trout.

CONCLUSIONS

The results of this landscape scale spatial analysis demonstrate that differences in the relative abundance of westslope cutthroat trout are associated with the cumulative effects of disturbance attributed to forestry operations. Significant relationships between the spatial distribution and abundance of cutthroat trout were detected using watershed disturbance variables as defined by the BC Interior Watershed Analysis Procedure, many of which have been employed by other salmonid research endeavours. Use of existing Fish and Fish Habitat Inventory datasets to determine relative cutthroat trout abundance yielded fish densities similar to other research, when used in concert with domain-expert input. The project results provide useful insights into landscape scale forest-fish interactions that indicate limitations to the conservation of westslope cutthroat trout. The findings of this project should be used in concert with available habitat knowledge at other scales (i.e. site level) to comprehensively address the conservation needs of cutthroat trout. The results clearly indicate the need for more comprehensive spatial analysis in designing effective policy for the protection of westslope cutthroat trout.

REFERENCES

Anderson, P. G. (1998, May 1-4, 1996.). Sediment generation from forestry operations and associated effects on aquatic ecosystems. Paper presented at the Forest-fish conference: land management practises affecting aquatic ecosystems, Calgary, Alberta.

Baxter, B. V., Frissell, C. A., & Hauer, F. R. (1999). Geomorphology, logging roads, and the distribution of bull trout spawning in a forested river basin: Implications for management and conservation. Transactions of the American Fisheries Society, 128(5), 854-867.

BCFisheries (2001). Reconnaissance (1:20,000) Fish and Fish Habitat Inventory: Standards and Procedures. Victoria, BC Fisheries**:** 170 pp.

BCMoF (1995b). Interior Watershed Assessment Procedure Guidebook. Forest Practices Code of British Columbia. Victoria, BC Ministry of Forests**:** 99 pp.

BCMoF (1995c). Riparian Management Zone Guidebook. Forest Practices Code of British Columbia. Victoria, BC Ministry of Forests**:** 68 pp.

Barrett, J. C., Alden, H. G., & Scholz, J. G. (1998). Geologic influences on the response of stream channels to timber-harvest related impacts. In M. K. Brewin & D. M. A. Monita (Eds.), Forest-fish Conference: Land management practices affecting aquatic ecosystems. Calgary, Alberta: Natural Resources Canada, Canadian Forest Service.

Bescheta, R. L., Frissell, C. A., Gresswell, R. E., Karr, J. R., Minshall, G. W., Perry, D. A., et al. (1995). Wildfire and Salvage Logging: Recommendations for Ecologically Sound Post-Fire Salvage Logging Management and Other Post-Fire Treatments on Federal Lands in the West, Accessed 2005 http://inr.oregonstate.edu/download/beschta_report_1995.pdf,

Bisson, P. A., Raphael, M. G., Foster, A. D., & Jones, L. L. C. (2002). Influence of site and landscape features on vertebrate assemblages in small streams. U S Forest Service Pacific Northwest Research Station General Technical Report PNW GTR(563), 61-72.

Carver, M. (2001). Riparian Forest Management for Protection of Aquatic Values: Literature Review and Synthesis. Nelson, BC, Prepared for: Forest Stewardship Council, BC Regional Standards Team**:** 48 p.

Costello, A. B. and E. Rubidge (2003). COSEWIC Status Report on Cutthroat trout (Oncorhynchus clarki spp.) **-** Submitted Draft. Vancouver, BC., Committee on the Status of Endangered Wildlife in Canada**:** 73 pp.

Creasey, J. R. (1998). Cumulative Effects and the Wellsite Approval Process. Faculty of Graduate Studies. Calgary, University of Calgary**:** 198pp.

Creque, S. M., Rutherford, E. S., & Zorn, T. G. (2005). Use of GIS-Derived Landscape-Scale Habitat Features to Explain Spatial Patterns of Fish Density in Mighigan Rivers. North American Journal of Fisheries Management, 25(3), 1411-1425.

Dunham, J.-B., Young, M.-K., Gresswell, R.-E., & Rieman, B.-E. (2003). Effects of fire on fish populations: Landscape perspectives on persistence of native fishes and non-native fish invasions. Forest Ecology and Management.

Dunnigan, J. L., Bennett, D. H., & Rieman, B. E. (1998). Effects of forest management on westslope cutthroat trout distribution and abundance in the coeur d'alene river system, Idaho, U.S.A. In M. K. Brewin & D. M. A. Monita (Eds.), Forest-fish Conference: Land management practices affecting aquatic ecosystems. Calgary, Alberta: Natural Resources Canada, Canadian Forest Service..

Dwire, K. A., & Kauffman, J. B. (2003). Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management, 178(1-2 SU -), 61-74..

Eaglin, G. S., & Hubert, W. A. (1993). Effects of logging and roads on substrate and trout in streams of the Medicine Bow National Forest, Wyoming. North American Journal of Fisheries Management, 13(4), 844-846.

ESRI 2002. ArcGIS Software version 8.1. Environmental Systems Research Institute, Inc. Redlands, California.

Fausch, K. D., Torgersen, C. E., Baxter, C. V., & Li, H. W. (2002). Landscapes to Riverscapes: Bridging the Gap between Research and Conservation of Stream Fishes. BioScience, 52(6), p. 483 - 498.

Frissell, C. A., Liss, W. J., Warren, C. E., & Hurley, M. D. (1986). A Hierarchical Framework for Stream Habitat Classification: Viewing Streams in a Watershed Context. Environmental Management, 10, 199-214.

Huntington, C. W. (1998). Streams and Salmonid Assemblages within Roaded and Unroaded Landscapes in the Clearwater River Sub-basin, Idaho. In M. K. Brewin & D. M. A. Monita (Eds.), Forest-fish Conference: Land management practices affecting aquatic ecosystems. Calgary, Alberta: Natural Resources Canada, Canadian Forest Service.

Karr, J. R., Rhodes, J. J., Minshall, G. W., Hauer, F. R., Bescheta, R. L., Frissell, C. A., et al. (2004). The Effects of Postfire Salvage Logging on Aquatic Ecosystems in the American West. Biologica, 54(11), 1029-1033.

King, R. S., Baker, M. E., Dennis F. Whigham, Donald E. Weller, Thomas E. Jordan, Paul F. Kazyak, et al. (2005). Spatial Considerations for Linking Watershed Land Cover to Ecological Indicators in Streams. Ecological Applications, 15(1), 137–153.

Kondolf, G. M., Piegay, H., & Landon, N. (2002). Channel response to increased and decreased bedload supply from land use change: contrasts between two catchments. Geomorphology, 45(1-2), 35-51.

Paul, A. J., & Post, J. R. (2001). Spatial distribution of native and nonnative salmonids in streams of the eastern slopes of the Canadian Rocky Mountains. Transactions of the American Fisheries Society, 130(3), 417-430.

Pess, G. R., Steel, M. E. A., Bilby, R. E., & Greenberg, H. M. (2002). Landscape charactaristics, land use and coho salmon (Oncorhynchus kisutch) abundance, Snohomish River, Wash., U.S.A. Canadian Journal of Fisheries and Aquatic Sciences, 59, 613-623.

Rieman, B., Peterson, J. T., Clayton, J., Howell, P., Thurow, R., Thompson, W., et al. (2001). Evaluation of potential effects of federal land management alternatives on trends of salmonids and their habitats in the interior Columbia River basin. Forest Ecology and Management, 153(1-3), 43-62.

Rubidge, E., Corbett, P., & Taylor, E. B. (2001). A molecular analysis of hybridization between native westslope cutthroat trout and introduced rainbow trout in southeastern British Columbia, Canada. Journal of Fish Biology, 59(Supplement A), 42-54.

Schiess, P., & Krogstad, F. (2004). Forest Harvest and Transportation. In T. G. Northcote & G. F. Hartman (Eds.), Fishes and Forestry: Worldwide Watershed Interactions and Management (pp. pp 195-215). Cornwall: Blackwell Publishing.

Schlosser, I. J., & Angermeier, P. L. (1995). Spatial Variation in Demographic Processes of Lotic Fishes: Conceptual Models, Empirical Evidence and Implications for Conservation. In J. L. Nielsen (Ed.), Evolution and the Aquatic Ecosystem: Defining Evolutionary Significant Units of Anadramous Salmonids in the Pacific Northwest (Vol. 17, pp. p. 392-401): American Fisheries Society Symposium.

Schwarz, C. J. (2005). "Stat-403/Stat-650 Intermediate Sampling and Experimental Design and Analysis." Retrieved December 20, 2005.

SPSS (2004). SPSS Statistical Software Release 13.0.1 12 Dec 2004.

Valdal, E. J. (2006). Cumulative Effects of Landscape Disturbance on Westslope Cutthroat Trout in the Upper Kootenay River Watershed: Implications for Management and Conservation. Environmental Science Program, Faculty of Environmental Design. Calgary, AB., University of Calgary**:** 97 pp.

Valdal, E. J. and M. Quinn, (2010). "Spatial Analysis of Forestry Related Disturbance on Westslope Cutthroat Trout in the Upper Kootenay River Watershed: Implications for Management and Conservation." Applied Spatial Analysis and Policy. Published Online Jan. 12 , 2010

Personal Communications

Brookes, S. C. (2003). Soils Scientist, Rocky Mountain Forest District, BC Ministry of Forests. Personal communication with E. J. Valdal. Invermere.

Chirico, A. (2003). Fisheries Information Specialist, Nelson Region, BC Ministry of Environment. Personal communication with E. Valdal. Nelson, BC.

Holmes, P. (2005). Forest Ecosystem Specialist, East Kootenays, BC Ministry of Environment. Personal communication with E. Valdal. Invermere, BC.

WILD TROUT RESTORATION IN ACIDIFIED STREAMS

Michael V. Shingleton and Walter S. Brown

Assistant Chief of Coldwater Fisheries and Stream Restoration Program Supervisor, West Virginia Division of Natural Resources, P.O. Box 67, Elkins, West Virginia 26241. 304-637-0245. mikeshingleton@wvdnr.gov, stevebrown@wvdnr.gov

ABSTRACT — An important goal of West Virginia Division of Natural Resources' Stream Restoration Program is to restore and maintain high-quality sport fisheries in streams and lakes that have been degraded by acid precipitation. Using limestone sand and drums, the restoration efforts of the program and its partners maintain sport fisheries in 392 miles of 85 acid-impacted streams and four lakes in 10 counties of the state. Over 10,000 tons of limestone are annually applied to state waters to sustain fisheries in both tributary and receiving streams. Approximately 80% of the stream mileage annually maintained under the program is managed principally for native brook trout Salvelinus fontinalis. Water quality and fish populations have improved in the restored streams. Increased angler expenditures on streams restored under the program greatly exceed the cost of restoration. Each restoration dollar spent by the state is estimated to produce US$47 of economic impact, most of which accrues to private sector businesses.

HISTORY

Up to 25% of trout streams in West Virginia have been severely degraded as a result of acid precipitation or acid mine drainage (Menendez et al. 1996). Many of these streams historically held populations of native brook trout *Salvelinus fontinalis*, West Virginia's only native salmonid. Degradation is manifested in damaged stream ecosystems, reduced populations of trout and other sensitive fish species, loss of angling opportunities, and economic losses from reduced recreational expenditures.

Acid source reduction has been a priority of both state and federal governments and has been effective in reducing atmospheric acid deposition. However, in many watersheds the soil's acid-neutralizing capability has been permanently depleted. In-stream acid neutralization is often the most cost effective and efficient method of restoring acid-degraded streams.

The West Virginia Division of Natural Resources (WVDNR) and the West Virginia Department of Environmental Protection (WVDEP) use three principal methods of in-stream limestone treatment to restore fisheries. The first method, and the oldest of the three, uses water-driven, self-feeding, limestone drum facilities (Figure 1) to grind limestone aggregate into a fine slurry that is continuously released into streams, neutralizing stream acidity as it dissolves. A second method, which evolved from the first method, uses fine limestone sand that is annually applied to streams (Figure 1), again neutralizing acidity as it dissolves. The third method, employed exclusively by the WVDEP, uses water powered, auger-fed in-stream Aquafix™ dosers to dispense calcium oxide pellets into streams, most of which are warmwater streams and thus will not be extensively addressed in this paper. All three technologies were developed in West Virginia.

The WVDNR and the WVDEP have partitioned the responsibility for restoring the state's acid-impacted streams. The WVDNR uses hunting and fishing license revenue, federal aid dollars, and investment income from legal settlements to fund treatment of waters impacted by acid precipitation. Agency policy prohibits the use of this revenue to treat acid mine drainage. Treatment of mining-induced acidification is the responsibility of WVDEP, which has statutory responsibility for mining and reclamation and receives substantial federal funds that can be used for that purpose. In watersheds that have been degraded by both acid precipitation and acid mine drainage, restoration projects involve partnerships between WVDNR and WVDEP. Other significant partners include the U.S. Forest Service, Trout Unlimited, and private land

Figure 1. Self-feeding limestone drum station (left) and limestone sand being applied to stream (right).

companies such as Plum Creek Timber Company and MeadWestvaco Corporation.

The WVDNR has conducted its research and management activities for acid stream restoration under two principal federal aid projects. For many years, the F-24-R *Infertile Streams Inventory* (WVDNR 1998, 1999), gathered water quality and fisheries data on acidified streams. These data were subsequently used to prioritize streams for limestone treatment implemented under the current F-40-D *Acid Water Neutralization* project. Because water quality restoration and physical habitat restoration are complementary strategies for restoration of impaired fisheries, these two components are now organized under the WVDNR's Stream Restoration Program. Program activities are closely coordinated with the agency's Coldwater and Warmwater Fisheries Management Programs. Acid stream restoration is a key brook trout restoration strategy for West Virginia and other states participating in the Eastern Brook Trout Joint Venture (EBTJV 2007)

West Virginians understand the magnitude of the acid stream problem in the state and strongly support efforts to restore these degraded natural resources. Because of our past successes, public demand is very high for additional restoration. In a 1997 public opinion survey, 75% of the randomly polled respondents said that acid stream restoration should receive even more emphasis from state government (Duda 1998). Respondents assigned that issue a higher priority than any other survey issue. That level of public support continues to this day.

Case Study of a Successful Restoration Project

Red Run is a 7.4-mi, 2nd order, moderate to high gradient tributary of Dry Fork of Cheat River, located in north-central WV. The stream arises at an elevation of 3,900 ft (msl), flows in a southwesterly direction and enters Dry Fork at an elevation of 1,800 ft (msl). Most of the 6,830-acre watershed lies within the Monongahela National Forest. Figure 2 shows declining pH values through the 1980's and early 1990's, typical of many poorly buffered native brook trout streams on the Forest. Approximately 2 mi above its confluence with Dry Fork, Red Run flows over a 20-ft high limestone outcrop where the pH increases to a level that allows brook trout survival and reproduction. Historically, no fish had been collected upstream of the falls. In August 1997, DNR began placing limestone sand into the upper reaches of Red Run. In September 1997, 95 native brook trout from a Dry Fork tributary were transferred to Red Run. Surveys conducted in July 1998 yielded brook trout below the falls, but none at a station above the falls, even though the pH had remained above 6. Brook trout were again collected from a Dry Fork tributary and stocked in Red Run, with the majority placed above the falls. Surveys conducted in 1999 at the same two locations surveyed in 1998, yielded an estimated standing crop of 15.1 lbs/acre (Figure 3: results from two survey stations, one below falls and one above falls, were combined to get average values).

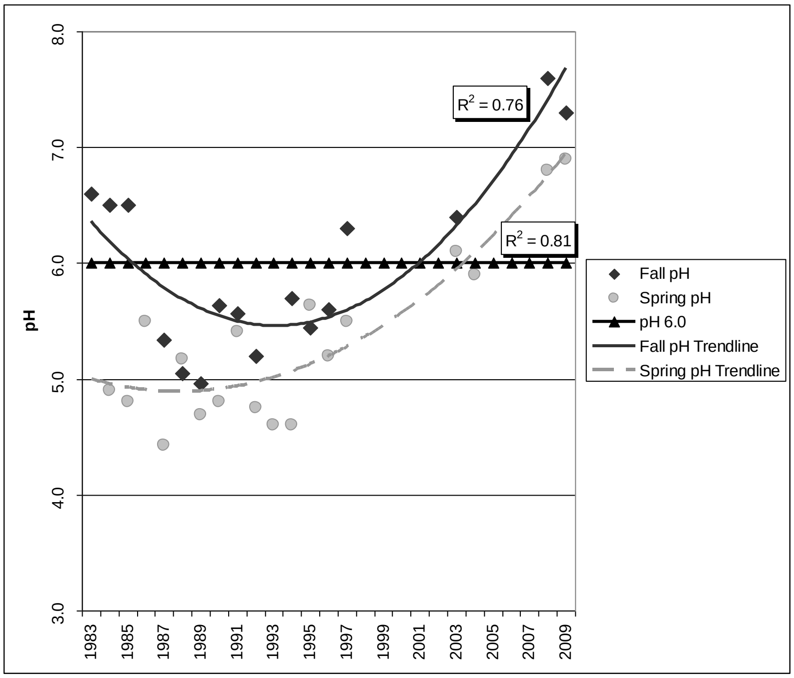


Figure 2. Red Run stream pH values and trendlines, spring and fall. Limestone sand treatment of Red Run began in August 1997. Data from WVDNR (1998), McClurg (2004), and Rebinski (2010)

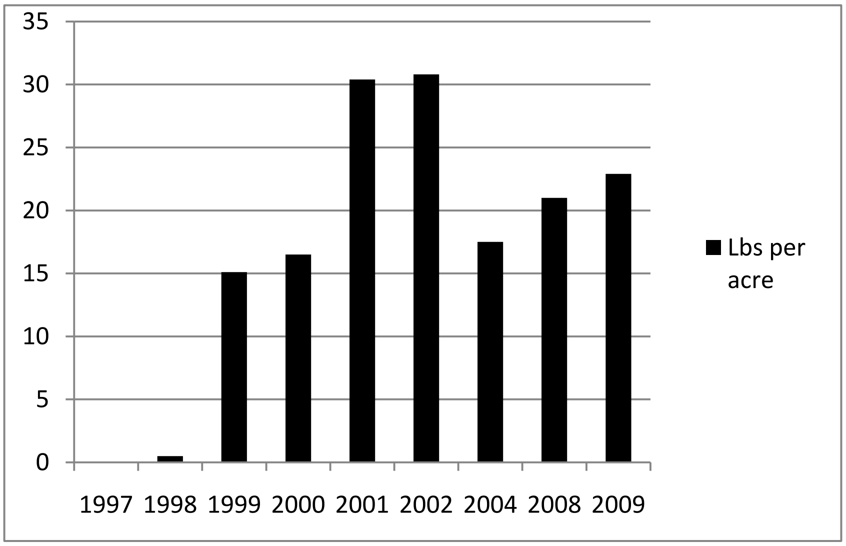


Figure 3. Red Run trout survey data: data averaged for survey stations below and above falls.

Although water quality improved immediately after initial treatment with limestone sand, the steep gradient (284 ft/mile) of Red Run, combined with the falls (Figure 4) resulted in the slow expansion of the brook trout population. However, after 2 years of treatment, standing crop, and the number of adult and juvenile brook trout did respond, and the population has expanded sufficiently to be one of West Virginia's better brook trout fisheries.

Figure 4. The falls of Red Run

Catch-and-release, fly-fishing-only regulations were implemented on Red Run in 1999, and continue today. This has likely helped the population to grow and thrive, and many anglers seek out the remoteness of the stream for successful fishing excursions.

Other Restoration Successes

Cranberry River

The Cranberry River is one of West Virginia's iconic trout streams. Many historical accounts exist of its extraordinary brook trout fishing. In the past quarter century, increased use and stream acidification from atmospheric deposition reduced fisheries management to a single option – catchable trout stocking. Even that option was threatened as acidification progressed. In 1989 and 1992, to neutralize stream acidity and restore management options, the WVDNR constructed two $1 million limestone drum facilities in the watershed. These facilities have been operating continuously since their installation. Water quality and fish populations have recovered to the extent that the watershed now provides 36 mi of high quality, semi-wilderness fishing in discrete management reaches that feature an array of wild and catchable trout angling opportunities.

Shavers Fork of the Cheat River

Another of West Virginia's most famous trout waters, the Shavers Fork has a storied history that includes turn-of-the-century visits by such luminaries as Thomas Edison and Henry Ford. Unusual for its high elevation, large river brook trout fishery, the Shavers Fork drew many such anglers to its historic Cheat Mountain Club, where photographs of record catches are still displayed. However, subsequent railroad construction, logging, and atmospheric acid deposition reduced its fishery to a seasonally stocked main stem with relic brook trout populations in the tributaries. By the 1970's, annual carryover of trout in much of the Shavers Fork main stem had ceased. Soon after, research began into the use of limestone sand to neutralize stream acidity. Today, through liming of 10 tributaries and the main stem, annual carryover of trout has been restored in 54 mi of the river. Recently, on the heels of successful water quality restoration, major efforts have begun to restore physical habitat in the river and tributaries to benefit brook trout. Projects are underway to eliminate fish passage barriers and enhance in-channel habitat diversity.

Blackwater River

The Blackwater River is another of West Virginia's well known trout fisheries. Once known as an exceptional brook trout fishery, the Blackwater River has been warmed and partially acidified as a result of historic logging and mining practices. By the 1960's, the canyon section of the river was essentially rendered fishless by this degradation of water quality. In 1996, the WVDEP and WVDNR partnered to construct a limestone drum facility on the main stem. The resulting water quality improvement has restored a mixed trout fishery to 12 mi of the river, including 3.5 mi of wild brown trout fishing in the spectacular and remote Blackwater Canyon.

North and South Forks of Cherry River

These two watersheds are drained by their respective main stems, and converge at the historic mill town of Richwood. Much of the North Fork watershed is owned by the U.S. Forest Service, while the South Fork watershed's largest landowner is Plum Creek Timber Company. Streams in both watersheds have been severely impacted by atmospheric acid deposition. To address this degradation, the WVDNR has partnered with the U.S. Forest Service and Plum Creek to restore both wild and stocked trout fisheries through limestone sand applications. Plum Creek has vigorously embraced this partnership, opening their lands to provide public fishing access, cost-sharing limestone sand applications, and modifying timber management practices to reduce stream sedimentation. These contributions have been recognized in their exceptional, independent audits conducted as part of the Sustainable Forestry Initiative.

Middle Fork River

The Middle Fork River watershed has benefitted from another unique partnership to restore water quality and fisheries. Degradation of water quality from mining and atmospheric acid deposition had impacted many of the watershed's fisheries and angling opportunities. In 1996, the WVDEP and WVDNR partnered to implement an extensive strategy addressing these problems. Twenty-nine headwater tributaries were treated and continue to be maintained by annual applications of limestone sand. As a result, brook trout populations in the tributaries have been bolstered and mixed trout fisheries in the main stem have been sustained. In all, 119 mi of trout fishery in the watershed have been restored.

The Statewide Program

Waters Treated Using Limestone Drums

The WVDNR currently operates two limestone drum stations on two streams in the remote Cranberry River Watershed. These backcountry facilities are administratively accessed by a gated service road in the spring, summer, and fall and by snowcat during the winter. The facilities are operated and maintained by two full-time employees. The combined consumption of the two stations is approximately 1,600 tons of high-calcium, limestone aggregate annually. The combined operating cost of the two stations in combination is approximately $133,000 annually. These stations have restored and now maintain high-quality trout fisheries in approximately 36 mi of stream.

The WVDEP constructed and currently funds the operation of another limestone drum facility on the Blackwater River. Under a cooperative agreement, this station is operated by WVDNR employees who are based at another location and periodically visit the station to perform maintenance. This station consumes approximately 2,000 tons of limestone annually, at an annual operating cost of approximately $125,000. It has restored and now maintains water quality in 12 mi of trout stream.

Combining figures for all three limestone drum stations and comparing the aggregate cost to the aggregate mileage of restored streams, the average annual cost per mile of restored stream is estimated to be approximately $5,375.

Waters Treated Using Limestone Sand

Currently, the WVDNR applies finely ground, high-calcium limestone sand to 53 streams in 10 counties of the state. Approximately 5,345 tons of limestone sand are applied to these streams each year to maintain water quality in an estimated 225 mi of restored trout streams. On a per mile basis, this equates to about 24 tons, or one truckload, of sand per mile of restored stream. The total program cost of these applications is approximately $153,000 annually.

In addition to in-stream application of limestone sand, the WVDNR applies 160 tons of limestone sand to 4 acidified lakes totaling 86 acres of restored fishery. The average annual cost per acre of these applications is about $54, with a total annual program cost of $4,600.

The WVDNR's treatment costs on some streams and lakes are shared with West Virginia Council of Trout Unlimited, Plum Creek Timber Company, MeadWestvaco Corporation, and the U.S. Forest Service. Significantly, the annual Trout Unlimited contribution to this program represents approximately 13% of the total program costs.

The WVDEP also funds major acid stream neutralization activities using limestone sand. Currently, the WVDEP funds the annual treatment of 119 mi of 29 streams at an annual cost of about $98,000. This is a collaborative effort with the WVDNR which provides most of the personnel time required each year.

The WVDNR's staff commitment to the overall limestone sand effort is approximately 1.5 FTE's per year. For that commitment, an estimated 344 mi of impaired stream fisheries and 86 acres of impaired lake fisheries have been restored and are now maintained. Most importantly, about 80% of the restored stream mileage is managed principally for native brook trout. Using limestone sand, our estimated average cost per mile of restored fishery is $730 per year.

Economic Impact of Restored Trout Fisheries

Considering both limestone drum treatment and limestone sand treatment, the Stream Restoration Program has restored and annually maintains water quality in an estimated 392 mi of restored trout streams. Apportioning the most recent estimates of angler use and economic impact to these 392 mi yields some significant conclusions. These restored waters would be expected to support an estimated 341,000 days of angling use annually. The estimated economic impact of recreational angling on these restored waters is expected to be $24 million annually. Most of the economic impact of angling accrues to private sector businesses providing equipment, supplies, food, lodging, and travel services. A cost/benefit ratio calculation for acid stream restoration under this program indicates that each dollar spent by the state produces about $47 of principally private sector benefit. Additional downstream benefits, both recreational and economic, accrue from the cumulative effect of extensive acid neutralization in upstream headwaters.

Conclusion

As stated earlier, WV's Stream Restoration Program has expanded from just treating water quality issues with limestone, to now include habitat restoration as well. Funding for current limestone sand placement and operation of the drum stations is fairly secure. New waters determined to be suitable limestone sand treatment are not added to the program unless additional long-term funding sources are available. This additional funding could come from more settlements by large power companies for air pollution issues.

Funding for habitat restoration has been secured to move these activities forward for several years. However, additional funding will be needed to make this portion of the program have meaning and benefits for the resource, as well as for future generations of anglers. Funding for habitat restoration will be sought through grants from federal and state agencies, and non-government organizations. Recognizing that a single funding source is unlikely to have sufficient funding to complete large projects, partnerships will be formed with various entities to secure the necessary funding.

There is still a great deal to do – but there is also great public and agency support to see it done.

Acknowledgements

The authors wish to acknowledge the pioneering research on acid stream neutralization conducted by WVDNR personnel Peter Zurbuch, Ray Menendez, Janet Clayton, Don Phares and others. Their work laid the groundwork for all of today's successes. Also gratefully acknowledged are the strong cooperative efforts of Trout Unlimited, Plum Creek Timber Company, MeadWestvaco Corporation, the U.S. Forest Service, and the West Virginia Department of Environmental Protection. Finally, the program owes a strong debt to current WVDNR employee John Rebinski, who continues to discover new restoration opportunities as he simultaneously maintains all of those that have already come into the program.

Literature Cited

Duda, M.D. 1998. West Virginia Residents' Attitudes Toward the Land Acquisition Program and Fish and Wildlife Management. Final report presented to the West Virginia Division of Natural Resources. Responsive Management, Harrisonburg, Virginia.

EBTJV (Eastern Brook Trout Joint Venture). 2007. Eastern brook trout: roadmap to restoration, (www.easternbrooktrout.org).

McClurg, S.E. 2004. Stream ecosystem response to mitigative limestone treatment in acid impaired, Central Appalachian Streams. M.S. Thesis. West Virginia University, Morgantown.

Menendez, R., J.L. Clayton, and P.E. Zurbuch. 1996. Chemical and fishery responses to mitigative liming of an acidic stream, Dogway Fork, West Virginia. Restoration Ecology 4(3): 220-233.

Rebinski, J. 2010. Personal communication, WVDNR.

WVDNR. 1998. Chemical investigations of acid precipitation impacted trout streams. *In* Acid water studies, Annual Performance Report; January 1, 1998 through December 31, 1998, Federal Aid Project F-24-R-17. West Virginia Division of Natural Resources, Charleston, WV.

WVDNR. 1999. Chemical investigations of acid precipitation impacted trout streams. *In* Acid water studies, Annual Performance Report; January 1, 1999 through December 31, 1999, Federal Aid Project F-24-R-18. West Virginia Division of Natural Resources, Charleston, WV.

RiverRAT: Tools for Analyzing Stream Engineering, Management, and Restoration Proposals

Peter Skidmore[1], Tim Beechie[2], Janine Castro[3], Brian Cluer[4], George Pess[5], Conor Shea[6], and Colin Thorne[7]

[1]Skidmore Restoration Consulting, Bozeman, MT, restoringrivers@yahoo.com;
[2]NOAA Fisheries, Seattle, Washington, tim.beechie@noaa.gov;
[3]US Fish and Wildlife Service, Portland, Oregon, Janine_M_Castro@fws.gov;
[4]NOAA Fisheries, Santa Rosa, California, brian.cluer@noaa.gov;
[5]NOAA Fisheries. Seattle, Washington, george.pess@noaa.gov;
[6]US Fish and Wildlife Service, Arcata, California, Conor_Shea@fws.gov;
[7]Professor, University of Nottingham, UK, colin.thorne@nottingham.ac.uk

ABSTRACT — Restoration and stabilization of rivers to restore trout habitat are now common practice. However, well-intentioned restoration efforts may be of little value or even lead to further degradation when management schemes do not accommodate underlying physical stream processes or broader watershed context. To foster more stream management and restoration projects, the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service collaboratively commissioned the development of River Restoration Analysis Tools (*RiverRAT)*. *RiverRAT* facilitates the critical evaluation of

a. watershed-scale biological, physical and social context;
b. the project planning process, including development of goals and objectives;
c. project design;
d. project implementation; and
e. compliance and effectiveness monitoring.

RiverRAT tools include:

1) *Project Screening Risk Matrix* - evaluates relative risk to aquatic resources related to project type and stream type;
2) *Proposal Checklist* - proposal completeness and information checklist; and
3) *River Restoration Analysis Tool (RiverRAT)* - web-based guide and project review tool for evaluation of any stream management or restoration proposal.

An accompanying *Science Document* and appendices synthesize concepts in fluvial geomorphology and river management, and present this information in an accessible format, particularly for those with predominantly biological background and expertise. *RiverRAT* resources are available at www.restorationreview.com and will be available through a NMFS website in 2010.

Background and Need

Management of stream corridors spans a wide range of intended outcomes, including stabilization of eroding stream banks, management or diversion of in-stream and flood flows, sediment management, river restoration and habitat enhancement to promote a species or biodiversity, or for mitigation. However, streams are complex and dynamic systems, and projects undertaken with the best of intentions may still cause unintended outcomes that could pose unacceptable risks to fisheries or habitat, either directly or by imposing additional constraints on natural processes. While implementation may result in short-term impacts, alteration of fluvial processes may result in longer-term, and thus more adverse, effects.

Guidelines and manuals currently exist for the development of specific elements of stream management projects; however, their focus is typically on the engineering or design aspects without provision for a watershed process or management context. No accepted standard of guidance exists for stream management projects; hence, all guidelines are limited in scope with respect to the specific needs of the reviewing regulatory agencies.

The National Marine Fisheries Service, a branch of the National Oceanic and Atmospheric Administration, and U.S. Fish and Wildlife Service (USFWS) (collectively referred to as the Services) have a responsibility to evaluate river projects funded, authorized, permitted, conducted, or consulted on in any way by the Federal agencies - in essence, any project in a river environment that may have an impact on protected species or the stream processes, habitat, or ecosystem they depend upon. Other federal and state agencies bear similar responsibilities for evaluating proposed stream projects in a range of specific regulatory contexts. All organizations that fund or permit stream projects have an inherent responsibility to evaluate projects and measure their success relative to stated goals and objectives. Our team has identified a specific need for tools that inform and guide the review or planning of river management projects in the context of both watershed setting and fluvial geomorphic processes. To this end, we have produced the *RiverRAT* (River Restoration and Analysis Tool) resources and training that create a solid scientific foundation for a thorough and comprehensive review of river restoration projects. *RiverRAT* resources are available online at www.restorationreview.com. Trainings are provided at request.

The goals of this project were to enable project reviewers to

- Understand the connections between physical processes and aquatic habitat;
- Understand the connection between common management actions, effects, and associated risks to protected species and habitat;
- Understand alternatives that can minimize project-related risks to protected species and habitat;
- Provide science and understanding that promote the design of sustainable projects, resilient to physical processes and changing environmental conditions;
- Document and streamline project review, and foster consistency among project reviewers; and
- Promote effective post-project appraisals, leading to more effective future river management.

While an emphasis on salmonid recovery and Endangered Species Act (ESA) context in the Pacific Northwest and California is inherent in this *RiverRAT*, the resources and tools have broader utility and have been easily adapted to other agencies' jurisdictions, other geographic regions, and specific ecological resources.

RiverRAT Resources

RiverRAT includes four resources:

1. *Science Document and Appendices* - synthesize major concepts in fluvial geomorphology and river management, thus providing a scientific foundation for evaluating potential impacts of stream projects, and providing a foundation for river management project planning;
2. *Project Screening Risk Matrix* evaluates relative risk to the aquatic resource due to project type and stream response potential and assists in determining appropriate level of review;
3. *Project Information Checklist* - indicates whether proposals include all information necessary to allow for a critical and thorough project evaluation; and
4. *River Restoration Analysis Tool (RiverRAT)* - guides reviewers through the steps necessary to critically evaluate the project planning process or to develop a stream management project.

The resources are designed to be stand-alone resources, but are also complementary. The *Risk Matrix* provides an initial screening tool for determining an appropriate level of review effort; the *Project Information Checklist* enables a quick assessment of the proposal completeness; and *RiverRAT* guides either a comprehensive review of a proposed project or the development of a project. The *Science Document* and *Appendices* serve as a primer and reference for all RiverRAT tools. Together, the document and tools provide a sound foundation in fluvial geomorphology and its relevance to river habitat so that proposed projects may be thoroughly evaluated in a timely manner with respect to their potential risks to species and habitat.

All print resources are available for download from www.restorationreview.com. While *RiverRAT* is a free online tool, it requires a user account obtained by request via email.

Science Document

The *RiverRAT Science Document* begins with a description of the three tools for project review. The bulk of the *Science Document* is then devoted to a synthesis of the integrated science of fluvial geomorphology as it relates to river habitat, starting with physical watershed controls, and progressing through stream processes and channel forms, thus providing a thorough scientific foundation for evaluating the potential impacts of stream projects. The document also presents a logical process for the development of engineering or management actions in rivers, including those intended to improve habitat, such as restoration and stabilization projects.

To facilitate deeper review of project design and analyses, the science document also includes:

Appendix 1- investigative analyses that form the basis for evaluating existing and proposed conditions, comparison of alternatives, and project design.

Appendix 2- design approaches and the application of design criteria to development of specific design elements as well as for developing specific monitoring metrics.

Appendix 3- additional stream corridor management alternatives.

Appendix 4- an annotated bibliography of stream management and restoration design guidelines.

The *Science Document* highlights common approaches to stream management (including restoration) that may not account for temporal or spatial variability or may actually constrain natural channel processes. Projects proposed as restoration, stabilization, or remediation often include project elements that are site-specific (e.g. 10's to 100's of meters in stream length), in large part because many constraints to aquatic species are identified at this scale. Many projects are unsuccessful, because they address local-scale symptoms without understanding the wider causes of habitat loss or degradation, which are often reach or watershed scale problems. Site-specific actions, such as meander reconstruction, the addition of weirs, installation of large wood structures, and bank stabilization, have become the default solution to many habitat problems and constraints, yet they are often planned and implemented without consideration of physical processes that may influence their outcomes or the potentially negative impacts of some project elements.

Application of traditional engineering design standards, such as 'factors of safety' biased towards structural stability, affords certain benefits in terms of professional accountability and rigorous analysis, but also simultaneously tends to increase risk aversion. The inherent problem with risk aversion in 'stream restoration' schemes is that it commonly leads to a greater reliance on engineered structures to ensure an acceptable 'factor of safety'. The resulting projects often impose unnecessary and undesirable constraints on natural channel adjustment and evolution, i.e., limiting long-term habitat value and potentially inhibiting habitat creation and maintenance.

To address these issues, the science document and tools facilitate identification and evaluation of the constraints, uncertainties, and risks associated with proposed projects. To this end, the document and tools encourage project development and review to include:

- Understanding how engineering and management actions affect the physical stream processes operating at varying scales (e.g., site, reach, and watershed);
- Accepting that uncertainty is inherent in all engineering and management actions in rivers with respect to predicting project outcomes and potential risks to physical processes and the habitats and species they sustain;
- Promoting solutions to identified problems that address the root causes at appropriate scales, rather than simply treating the symptoms of the problem at the site-scale; and
- Acknowledging that human influences are fundamental components of all ecosystems, at all scales.

RiverRAT Project Screening Risk Matrix

Effective and efficient review of stream projects begins with a determination of relative project risk that can facilitate the determination of an appropriate level of effort in review. To help reviewers develop and improve their capability to match the intensity and extent of review to the inherent project risk, a screening tool has been developed. The *Project Screening Risk Matrix* (Figure 1) evaluates risk to resource due to stream response potential (X-axis) and risk to resource due to project impact potential (Y-axis). In screening out low risk projects on low risk streams and using the time saved to allow deeper scrutiny of higher risk projects and more sensitive streams, responsibility for balancing expediency against thoroughness rests with the individual.

The principle underlying the *Risk Matrix* is that actions and projects should do no lasting harm to the resource. Within this principle, reviewers will assess the risk of doing harm to ‘resource’ within the context of the relevant legislation. For example, in the case of NMFS this will usually center on Section 7 of the ESA, and so ‘resource’ will refer to one or more listed species and their habitat. The U.S. Army Corps of Engineers (USACE), which operates under Section 404 of the Clean Water Act, would have an expanded definition of ‘resource’, which in this case refers to a ‘Water of the United States’. The *Risk Matrix* as presented here may be adapted for use by different reviewers and agencies and in different contexts according to their needs.

The *x-axis*, Stream and Site Response Potential, represents the risk to resource associated with the sensitivity to natural or anthropogenic disturbance of the stream and its habitat. This axis uses stream attributes, such as gradient, bed and bank material, and localized geomorphic context, to assist reviewers in making an initial assessment of the overall risk to resource stemming from the landscape context, natural system resiliency, and imposed human modifications. Some stream types are naturally sensitive to disturbance, while others may have become sensitized due to land use history and past engineering and management in the river network.

The *y-axis*, Project Impact Potential, represents the risk to resource associated with the proposed action or project type. Some disturbance to the fluvial system is inevitable when performing actions in or near a stream or undertaking a restoration scheme. This axis, therefore, uses indicators of the project scale, context, cumulative impacts, introduced artificial constraints, and the ability to detect impacts to assist reviewers in making an initial risk assessment of the proposed action or project.

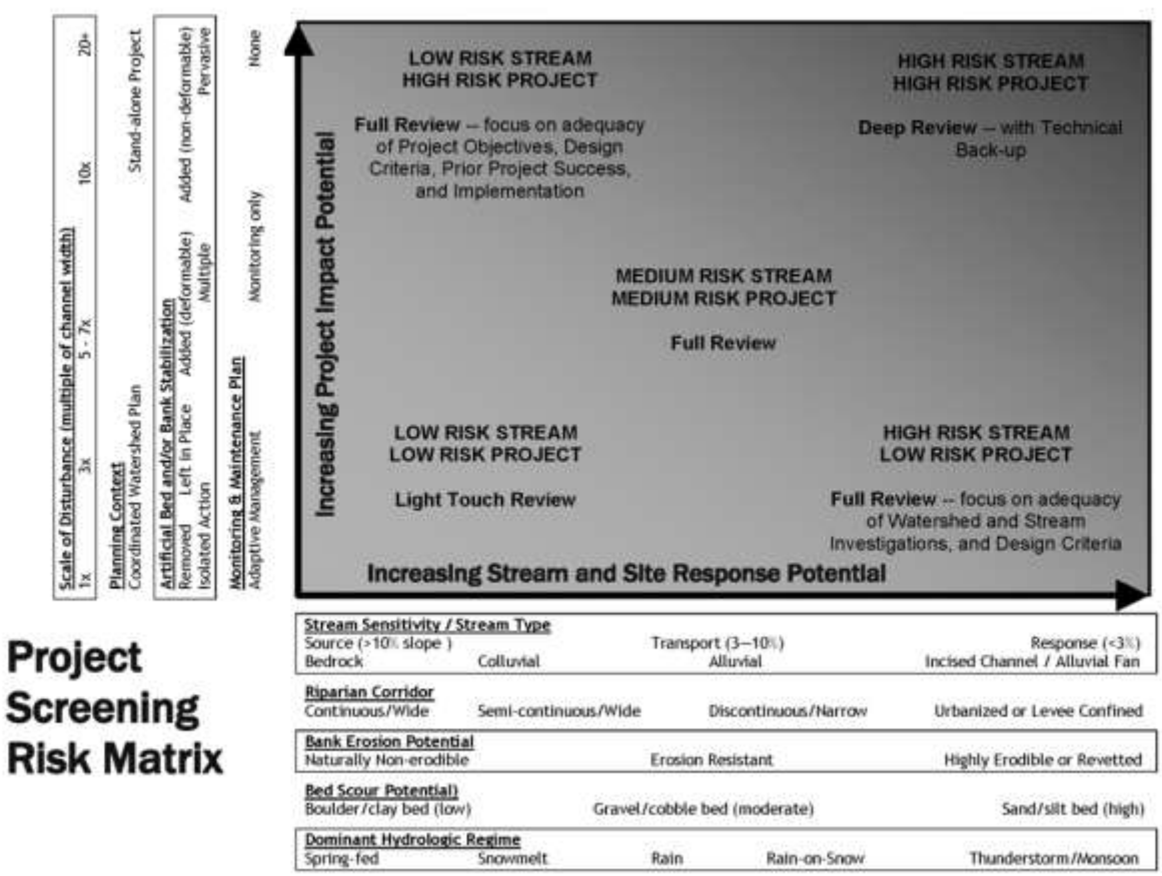


Figure 1. The Project Screening Risk Matrix

The overall risk to resource is represented only by a gradient (Table 1) that begins with light gray for low risk (lower left) and ranges to dark gray for high risk (upper right) in the risk matrix. The axes of the risk matrix presented here purposefully have no scales; similarly, no examples are given of projects that might typify a particular level of impact potential or streams that possess representative levels of response potential. The lack of quantification and examples does not reflect a lack of knowledge or understanding of potential project impact and stream response. Rather, the matrix has not been quantified or populated because there is no cook-book way to assess the risks associated with a proposed action or project *a priori* and without context. Our purpose here is not to tell end-users the answers to difficult questions, but to help them to understand risks and pose the right questions in the first place.

RiverRAT Project Information Checklist

The *RiverRAT Project Information Checklist* queries the user regarding project information sufficiency and applicability. It includes a comprehensive list of all information that a project proposal could contain for a thorough review by Services' staff and has been developed for use as a template for a Biological Assessment (BA), thus providing a consistent model for the organization and content of a complete BA. An excerpt of the detailed checklist is provided in Figure 2.

The primary purpose of the *Checklist* is to determine if there is sufficient information provided to facilitate the use of *RiverRAT*. However, it is provided as an open source file and can be catered to any specific application.

By providing all information suggested in the checklist, a project team can avoid delays during the review process, and a reviewer can be reasonably assured that a project team has put in the effort required to develop a well thought-out project that encompasses appropriate spatial and temporal scales, landscape context, risk, design approach, and adaptive management. Ideally, use of the checklist by both project developers and reviewers will promote time and resource efficiency and will make the review and consultation process more transparent to both parties.

Table 1. Selection of review effort based on risk sources and level of overall risk to resource

Impact & Response Potential	Risk to Resource	Indicated Treatment
Low Risk Project Low Risk Stream	Low	As both sources of risk to resource associated with this action or project are low, overall risk is low and a light touch may be taken in evaluating it using *RiverRAT*.
High Risk Project Low Risk Stream	Medium	As the action or project carries a high risk to resource the proposal merits a full review, paying particular attention to the adequacy of: the Project Objectives, those Elements of the project that pose the greatest threats, the Design Criteria, evidence of prior success with similar projects, and the implementation plan. However, as the risks associated with the stream are low, it is likely that responses to the action or project will be limited to the project and adjacent reaches. Hence, a lighter touch may be taken in evaluating the wider watershed and stream system contexts and implications of the proposed work.
Medium Risk Project Medium Risk Stream	Medium	As risks arise equally from the project and the stream in which it is to be implemented, a full review, involving careful application of *RiverRAT*, should be performed.
High Risk Stream Low Risk Project	Medium	A low risk project may still pose serious risk to resource if implemented in a stream that is highly sensitive to destabilization. Hence, a full review is merited, emphasizing scrutiny of the adequacy of Watershed and Stream Investigations, Design Criteria related to preventing the impacts of the project from perturbing the fluvial system and plans for post-project monitoring and adaptive management to limit the effects of unforeseen impacts to within the project reach.
High Risk Project High Risk Stream	High	Proposals that have a high overall risk to resource merit a thorough review that goes beyond that routinely applied. Proposals in this category are often complicated or ground breaking and it may also be the case that the engineering, geomorphological or other technical/social/economic aspects of the proposal are sufficiently complex or challenging as to require back-up from subject specialists, and reviewers should not hesitate to seek assistance where necessary.

	Y	N	NA	
	Y	**N**	**NA**	***Design team***
57	☐	☐	☐	Name and titles of firms and individuals responsible for design.
58	☐	☐	☐	List of project elements that have been designed by a licensed Professional Engineer.
	Y	**N**	**NA**	***Hydrologic analysis***
59	☐	☐	☐	Description of historic, ongoing, or anticipated impacts to basin hydrologic regime.
60	☐	☐	☐	Summary of hydrologic analyses conducted, including data sources and period of record.
61	☐	☐	☐	List design discharge (Q) and return interval (RI) for each design element.
	Y	**N**	**NA**	***Sediment transport and dynamics analysis***
62	☐	☐	☐	Description of previous or anticipated impacts to basin or reach sediment supply.
63	☐	☐	☐	Summary of sediment supply and transport analyses conducted, including data sources.
64	☐	☐	☐	Describe sediment size gradation used in streambed design.
	Y	**N**	**NA**	***Hydraulic analysis***
65	☐	☐	☐	Summary of hydraulic modeling or analyses conducted and data source.
66	☐	☐	☐	Inundation map for design and flood flows before and after implementation.
	Y	**N**	**NA**	***Vegetation design***
67	☐	☐	☐	Species list, materials sources, and plant form.
68	☐	☐	☐	Planting plan map (distribution and density by species) and irrigation plan.
	Y	**N**	**NA**	***Soils and geotechnical analysis***
69	☐	☐	☐	Summary of geotechnical analyses including stratigraphy and grain size of materials.
70	☐	☐	☐	Groundwater elevation, flow direction and seasonality within floodplain and banks.

Figure 2. An example of details in the *Checklist*

RiverRAT – River Restoration Analysis Tool

RiverRAT is an on-line framework for project evaluation that guides reviewers through a thorough review of a project proposal (Figure 3). The entire project development process is addressed, beginning with problem identification, progressing through the design phase, and culminating with project monitoring. *RiverRAT* enables a review of project and design integrity with respect to species or ecosystem recovery.

It can be used in various regulatory review contexts, or by project proponents to guide project development and design. Access to *RiverRAT* by project sponsors, stakeholders, and specialists will guide them to developing project proposal documents that are both more informative and better tuned to the needs of regulatory staff who must review the proposal.

RiverRAT provides a review framework and links to additional technical resources and assistance that may be needed to support in-depth and detailed scientifically based and objective treatment that is justified for projects that carry a high risk to resource. Once logged in, the review tool steps the user through a series of 16 questions (Figure 3). Each question includes supporting information to help the user thoroughly evaluate each question in the proper context as well as a reference to the actual supporting document where the topic is thoroughly discussed.

SUMMARY

Our team produced a suite of tools, supported by scientific synthesis, for analyzing and developing river management and restoration projects and proposals. *RiverRAT* and its supporting tools, the *Risk Matrix* and the *Project Information Checklist,* help determine the depth of review required, ensure that a project proposal is complete, and guide reviewers through a thorough and scientifically sound project review. The tools can also be used to guide project proponents through a comprehensive project development process. The tools are supported by the *RiverRAT Science Document* **-** the scientific underpinning of the tools. Utilizing these tools can improve review consistency and transparency, and we believe that there can be a feedback with project development to improve project designs, and most importantly, place problems and solutions in context with physical process drivers and geomorphic controls on aquatic habitat creation and maintenance.

The goals of this project were to enable project reviewers to

- Understand the connections between physical processes and aquatic habitat;
- Understand the connection between common management actions, effects, and associated risks to protected species and habitat;
- Understand alternatives that can minimize project-related risks to protected species and habitat;
- Provide science and understanding that promote the development and design of sustainable projects, resilient to physical processes and changing environmental conditions;
- Document and streamline project development and review, and foster consistency among project reviewers; and,
- Promote effective post-project appraisals, leading to more effective future river management.

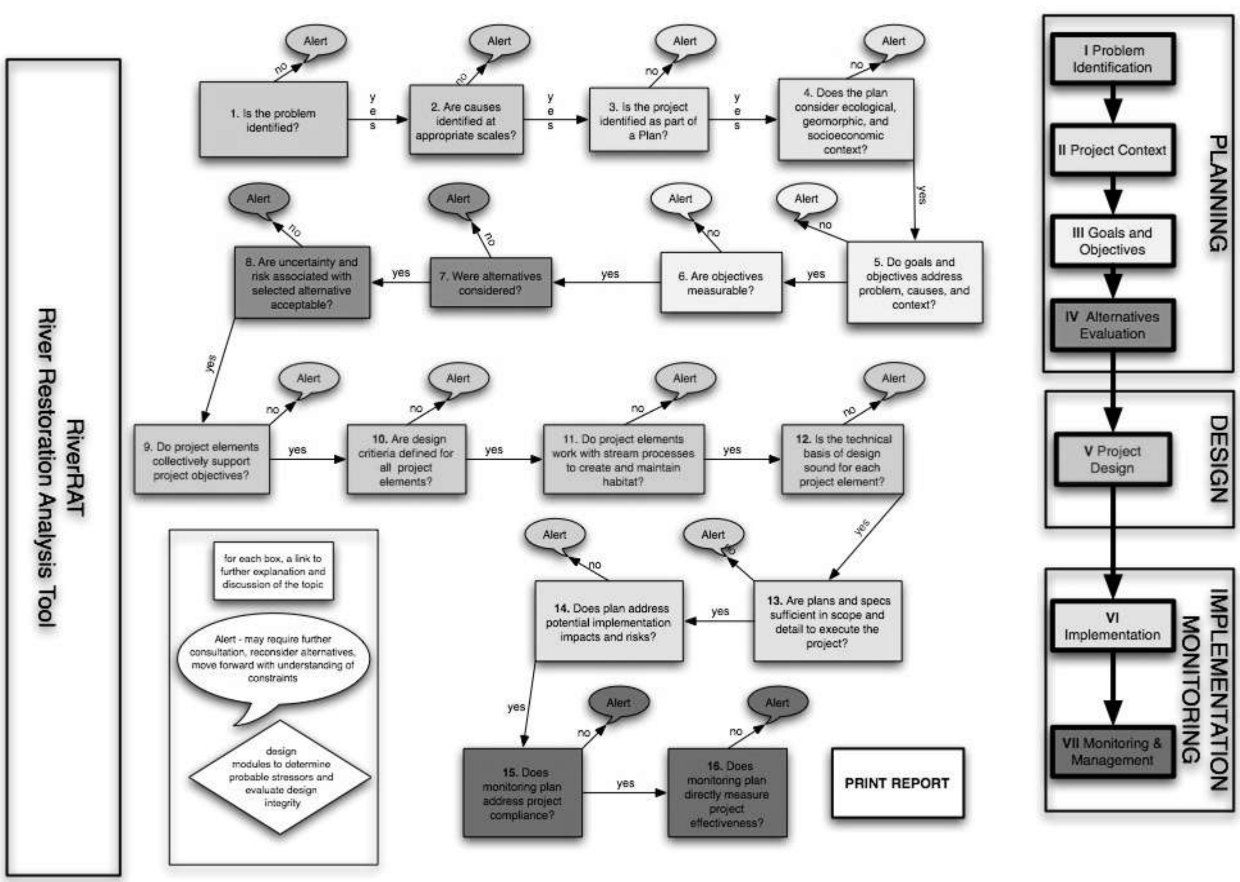


Figure 3. The *RiverRAT* framework.

ACKNOWLEDGMENTS

The RiverRAT resources were developed by a team of NMFS and USFWS staff, together with their contractors. In addition to the principal author team and peer review, a panel of experts was convened in December of 2007 for brainstorming the project and giving guidance, and who later reviewed drafts of the document. The expert panel included William Dietrich, University of California Berkeley, Peter Downs, Stillwater Sciences, Matt Kondolf, University of California Berkeley, Greg Koonce, InterFluve, Inc., and Douglas Shields, USDA-ARS National Sedimentation Laboratory. Additionally, interviews with Services managers, and workshops with over 50 potential end users from a wide range of state and federal resource agencies were conducted to solicit input, guidance, and feedback on draft products.

MEADE PEAK RANCH: A SUSTAINABLE MODEL FOR CONSERVING YELLOWSTONE CUTTHROAT TROUT, STREAM ECOSYSTEMS, AND TRADITIONAL LIVESTOCK RANCHING

Rick Williams[1], Bill Platts[2], Wayne Burkhardt[3], and Alan Prouty[4]

[1]Clear Creek Genetics, 524 W. Two Rivers Dr. Eagle Idaho 83616, troutdna@cableone.net, 208-938-9004

[2]Platts Consulting, 3920 Hillcrest Dr. Boise, ID 83704, bplatts@ecosystemsciences.com, 208-342-2505

[3]Ranges West, 2410 Little Weiser Rd., Indian Valley, ID 83632, rangeswest@ctcweb.net, 208-256-4437

[4]Director, Environmental and Regulatory Affairs, J.R, Simplot Co., 999 Main St., Suite 1300, Boise, ID 83702, Alan.Prouty@simplot.com, 208-336-2110

ABSTRACT — Meade Peak Ranch, previously, the Hartman Ranch, is located on the Idaho-Wyoming border and is transected by Crow Creek, a meandering tributary stream system of the Salt River, a stronghold for Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*. Meade Peak Ranch was acquired by the J. R. Simplot Company in 2006. The ranch has a long history as a working cattle ranch with bottomland meadow haying and upland grazing components including approximately 562 acres of irrigated meadows and 4,517 acres of upland range. Grazing impacts have affected much of the existing riparian zones on the ranch and on adjacent portions of Crow and Sage creeks. Since 2006, we have worked with the JR Simplot Company to develop a ranch and grazing management plan that will establish and maintain a healthy, functioning Ranch that also benefits biodiversity, special status species, improved water quality and livestock use efficiency, while providing for a sustainable ranching operation. To accomplish this, the entire Crow Creek riparian corridor on the ranch was fenced in 2007, creating a livestock-free buffer zone around the meandering stream course. In the spring of 2010, we completed the Meade Peak Ranch Management and Grazing Plan (MP-MGP), that will guide activities on the ranch to balance fisheries conservation and stream restoration goals with the continuation of a working ranch and its haying and livestock grazing components, so that conservation objectives and fisheries benefits are achieved sustainably while allowing the continued and economical operation of a working ranch. Here we present the results of this 4-year project as a potential model framework that might be used to balance fisheries and conservation goals within the context of a modern working ranch, rather than the common view that ranching and conservation goals are inherently incompatible.

Meade Peak Ranch on the Idaho-Wyoming border was acquired by the J. R. Simplot Company in 2006. The ranch has a long history as a working cattle ranch with 562 acres of irrigated meadows and 4,517 acres of upland range. Grazing impacts have damaged the existing riparian zones on the ranch and on adjacent portions of Crow and Sage creeks. Crow Creek, a tributary of the Salt River, is a stronghold for Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* and transects the ranch. We developed a ranch and grazing management plan to restore and protect degraded riparian systems, improve water quality, benefit Yellowstone cutthroat trout, and increase livestock grazing efficiency using a rest-rotation pasture system. The Meade Peak Ranch Management Plan will guide activities on the ranch so that conservation objectives and fisheries benefits can be achieved sustainably while allowing the continued and economical operation of the ranch. This approach can serve as a model ranch framework to balance fisheries and conservation goals within the context of a modern working ranch, rather than the common view that ranching and conservation goals are inherently incompatible.

An Opportunity Appears

Meade Peak Ranch was acquired by the J. R. Simplot Company in 2006. An initial inspection of the ranch revealed that Crow Creek flowing through the ranch had potential for riparian and stream restoration that would enhance already existing fisheries benefits. Williams and Williams (2006) provided a brief review of the ranch property, stream and fishery condition, and made several recommendations for improvement of the riparian corridor.

The Meade Peak Ranch (hereafter the Ranch) lies in the midst of a 10- to15-mile-long valley bottom transected by a meandering stream. This long valley bottom is entirely encompassed by a series of privately owned ranches; consequently, public access to the stream in these reaches is extremely limited. Grazing has affected much of the riparian zone throughout the drainage, with detrimental impacts differing widely from property to property.

Williams and Williams (2006) recommended fencing the Crow Creek riparian corridor within the Ranch to control livestock movement and limit further damage to stream banks and riparian vegetation. Additionally, sediment deposition in Crow Creek was high and exotic weeds, some of them noxious, were established in several places. Despite these conditions, Crow Creek still displayed high sinuosity, with little channel down cutting, and abundant Yellowstone cutthroat trout and brown trout *Salmo trutta* populations.

In 2007, the entire Crow Creek riparian corridor was fenced and the J. R. Simplot Company contracted with the authors (Williams, Platt, and Burkhardt) to develop a ranch management plan that sustainably balanced fisheries, wildlife, and conservation goals with a modern working livestock ranch.

Development of a Vision and Goals for the Ranch

During 2007, we developed a vision and goals for the Meade Peak Ranch that attempted to balance fisheries, wildlife, and conservation goals with operation of a modern working livestock ranch.

Ranch Vision and Management Goal

The ranch vision is to improve fisheries and wildlife habitats in concert with operating a "working ranch" including its haying and livestock grazing components. The ranch management goal is to achieve conservation objectives and fisheries benefits while continuing the economical operation of a "working ranch".

Grazing Management Goal

The grazing management goal is to establish a functioning ranch that benefits livestock use efficiency and ensures a sustainable ranching operation while benefiting biodiversity, special status species, and improved water quality. This goal will be achieved by developing a 5-year land and livestock grazing management plan that implements grazing "Best Management Practices" (BMPs). Adaptive management will be used to monitor, evaluate, and maintain a healthy economic ranching operation that meets stream and fisheries conservation goals. The plan will be revised every 3 to5 years incorporating amendments. The plan builds on principles and concepts articulated in Platts's (1991) examination of livestock grazing and its effects on salmonid fishes and stream habitats.

Scientific Foundation for the Ranch Management Plan

Cattle grazing has been a dominant land use throughout much of the western United States for well over a century. The impacts of cattle grazing on the landscape are widespread and well documented. Grazing impacts disproportionally affect riparian ecosystems compared to upland or rangeland habitats. Poorly managed cattle congregate in riparian areas to take advantage of shade, streamside forage, and access to water (Kauffman and Krueger 1984; Armour et al. 1991). This causes damage to the riparian system including overgrazing, decreased stream bank stability, increased sedimentation, decreased pool depths, and increased water temperatures. In turn, these impacts are thought to make the habitat less suitable for fish, resulting in eventual reduction in trout production (Platts 1991).

Current BMPs for riparian and stream management focus on preserving (or restoring as needed) stream bank stability. Managing the season, intensity, and duration of grazing in and around riparian systems will maintain (or restore) plant vigor and root biomass, which promote stream bank stabilization (Clary and Kruse 2004). Stable stream banks lead to deep pools, clean gravels, and promote complex instream structures, which provide suitable spawning, rearing, and wintering habitat for trout (Platts 1991). Several case studies have documented rapid increases in trout populations within 5 years in stream sections where cattle fencing exclosures were constructed to restore severely overgrazed stream sections (Knapp and Matthews 1996). Saunders and Fausch (2007) examined five pairs of streams in western Wyoming that had riparian zones under high-density short-duration (HDSD) grazing or season-long (SL) grazing treatments. Stream reaches under the HDSD grazing treatment had 2 to 3 times more riparian vegetation biomass and input of terrestrial invertebrates than stream reaches under SL grazing. Total trout biomass in HDSD stream reaches was more than twice that in SL reaches. These results suggest that improved grazing management practices have the potential to positively influence fish populations in western rangeland streams.

Similarly, BMPs for livestock management predict that protecting stream systems coupled with off-site watering and supplementation sources will lead to increased cattle production. McInnis and McIver (2001) found that off-stream water and salt attracted cattle into the uplands to forage, which reduced stream bank degradation from 9% in non-supplemented pastures to 3% in supplemented pastures. Cows and calves gained 0.27 kg/d and 0.14 kg/d, respectively, more in pastures with off-stream water and supplements than those in pastures without off-stream water and supplements (Porath et al. 2002).

Assessment of Conditions on the Ranch

Ranch Description

The Ranch includes 4,517 acres of upland range and 562 acres of irrigated bottomland meadows (Figure 1). The upland range is separated into two large grazing units, a north unit (3,467 acres) and a south unit (1,050 acres). The north unit is subdivided into two pastures of approximately equal size, a north pasture and a south pasture.

The upland range contains the steeper, mountainous terrain lying to either side of the Crow Creek Valley. The upland range units provide excellent summer pasture for livestock that when grazed under proper management, cause few adverse impacts to other resource values. Livestock grazing on Ranch upland rangelands is limited by the steep topography

Figure 1. Map of the bottomlands of Meade Peak Ranch along Crow Creek, showing the meandering sinuous nature of the creek as well as the three man-made ponds. Photo shows the upper part of Crow Creek and the meandering nature of the stream and the large (9.9 acre) upper Ranch Pond. Note grazing impacts on the east side in the Crow Creek pasture in the foreground versus the thick willow stands on the roadside where grazing is largely absent.

and limited watering sources. This has resulted in excessive grazing on more accessible areas around water sources. Little or no use occurs on the less accessible range areas.

Bottomland pastures bordering Crow Creek are productive, yielding high quality hay and forage. Pastures are fertilized as needed to increase productivity for haying and livestock grazing. Livestock water is abundant and well distributed in all pastures. Forage use by livestock within the riparian Corridor Parcel has been discontinued to achieve stream and fisheries conservation goals. All pastures are summer hayed, including the Corridor Parcel. Invasive weed control is practiced in all pastures.

Stream Conditions

The Ranch has 3.5 mi of stream channel. Four permanent flowing streams (Crow, Sage, Rock, and Bull creeks) flow into or initiate within the Ranch. Wetlands are associated with main Crow Creek, as well as with Rock Creek and Bull Creek (Figure 1). Streams on the Ranch are very productive because they drain watersheds with productive soils; consequently, they have high fish and wildlife potential.

Fish and Wildlife Status

The Ranch supports a high diversity of wildlife, ranging from Yellowstone cutthroat (YCT) and brown trout to mule deer, elk, sandhill cranes, beaver, pelicans, and breeding neo-tropical passerine birds in the riparian corridor. Yellowstone cutthroat trout are a species of "special concern" in the intermountain West (Meyer and Lamansky 2004; IDFG 2007). The Salt River contains the second highest number of YCT populations in Idaho. In the Crow Creek drainage, 34 stream mi contain YCT including 14.5 mi in the Sage Creek drainage. The IDFG's goals are to ensure the long-term persistence of YCT within its current range and to strengthen and expand YCT populations within their historical range. These goals are consistent with the Ranch's management goals.

Data from NewFields Boulder and HabiTech (2008) show brown trout and YCT using Sage and Crow creeks in the same ratios throughout the upper, middle, and lower reaches on the Ranch. Brown trout were about 3 times more common than cutthroat throughout, though sampling at the lower creek survey site yielded ~50% more fish of both species than the upper site (Table 1). At each sampling site in the Ranch, Yellowstone cutthroat trout were larger on average than brown trout. Average size for both species was larger at the downstream sampling site as compared to the upper sampling site.

Fisheries management goals on the Ranch are to develop higher quality and more diverse stream and riparian habitats along Crow, Rock, Sage, and Bull creeks. Much of this will occur naturally as a result of eliminating or reducing long-standing grazing impacts to stream banks and riparian systems. As stream banks revegetate and shrubs provide shade and cover, stream habitats will diversify, resulting in more pools, cover and lower water temperatures. This should benefit fish in the Ranch's aquatic systems. Fish populations will be monitored annually (NewFields Boulder and HabiTech 2008).

Current Land Use and Status

Since acquiring the Ranch in 2006, the J.R. Simplot Company has leased the upland rangelands northwest of Crow Creek Road to R.A. Peterson and Sons for livestock grazing. Petersons own adjoining rangelands to the north and west of the Meade Peak Ranch. In 2006, Petersons grazed 277 cow-calf pairs on the ranch from late May through September for approximately 1,108 AUMs (Animal Units per Month). This created grazing distribution problems that both Petersons and the ranch manager recognized. The steep terrain, limited stock watering sources, long grazing season and class of livestock

Table 1. A comparison of abundance and size of brown trout and Yellowstone cutthroat trout in Crow Creek sampling sites (in 2006) located in the upper and lower portions of the Meade Peak Ranch (NewFields Boulder and HabiTech 2008).

	Brown trout		Yellowstone Cutthroat Trout	
Sampling Site	Number Observed	Average Size	Number Observed	Average Size
Upper/Middle	135	203 mm	46	295 mm
Lower	202	232 mm	69	318 mm

all combined to cause excessive grazing use on the more accessible areas around watering sources. Little or no use occurred on the less accessible upland range areas.

In 2007, Petersons grazed 650 light yearlings from late May to early August for approximately 1,220 AUMs. Yearling cattle distributed grazing use across the North Unit better than the cow-calf pairs did. No upland range areas inspected showed significant levels of grazing use; however, utilization ranged from heavy to severe on the areas close to watering sources with moderate utilization on less accessible area further away from watering sources.

Prescriptions to Improve Conditions

The dominant impact across the Ranch has been the long-term effect of livestock grazing. Impacts were generally light in the uplands, but ranged from moderate in the bottomlands of Crow Creek to severe in the riparian zones of tributary systems. We used a tiered approach to improve habitat and grazing conditions on the Ranch. We first protected or preserved existing high quality habitats, then worked to repair and restore damaged habitats. Finally, we worked to enhance existing habitat features.

The most obvious example of protection and preservation on the Ranch was the construction of the riparian grazing exclosure fence along the entire Crow Creek riparian corridor. Crow Creek on the upstream third of the Ranch was already protected by a fencing exclosure constructed prior to 2006 by the previous landowner (Figure 2), with the remainder of Crow Creek fenced in 2007. Fencing the stream corridor will allow development of riparian grasses, sedges, and willows, which will further stabilize stream banks, thereby reducing sediment input and increasing streamside shading and cover.

In contrast to Crow Creek, Bull Creek (a tributary to Crow Creek on the Ranch) had been severely impacted by grazing for decades. An extensive fencing project occurred in 2010 where the lower 1-mi section of Bull Creek was fenced. The project included separate fencing of the intermediate and headwater springs, both of which had been severely impacted by livestock damage, as well as development of off-site and gap watering stations to provide water sources for livestock away from the recovering riparian system (Figure 3). As riparian and wetland areas recover, the water retention ability of the habitat should improve, and in the case of Lower Bull Creek, the water table should rise and wetlands and emergent grasses and sedges should spread from the enclosed riparian areas into the adjacent pasture edges where livestock will be able to eat them.

As Bull Creek recovers and rebuilds, improvements in channel depth, water temperature, water quality, and streamside vegetation that provide shading and cover, the stream will be capable of supporting spawning and juvenile rearing by Yellowstone cutthroat trout (Platts 1991; Saunders and Fausch 2007). Finally, the fence constructed for the Bull Creek riparian corridor is a wildlife-friendly fence that allows passage by deer, elk, and moose, but restricts cattle movement.

Figure 2. Fencing exclosure in the bottomlands of Meade Peak Ranch along Crow Creek in the upper portion of the Ranch, showing the sinuous nature and recovering riparian zone of the creek.

Figure 3. Bull Creek water sources. Off-site water sources (tires) adjacent to the fenced riparian corridor below the intermediate spring and a gravel lined 80-ft water gap located near the mid-point of the 1-mi long riparian corridor along Bull Creek northwest of the Crow Creek Road.

Our approach to habitat improvement and restoration on the Ranch has largely been a passive and natural one of attempting to remove or control negative impacts to the stream and riparian system, thereby allowing natural processes to occur and for the stream systems to repair themselves. Little active habitat manipulation has occurred at this point.

Development of the Meade Peak Ranch Management Plan

In 2009, it became apparent that an overall management plan was needed for the Meade Peak Ranch that would coordinate ongoing (and future) conservation activities with the ongoing livestock grazing operation. We recently completed that plan (Williams et al 2010). The MP Ranch Management Plan has three main components: a Grazing Plan, a Conservation Plan, and a Recreation Plan.

Grazing Plan

The Grazing Plan's management goal is to establish a functioning ranch that promotes trout production, biodiversity, water quality, and livestock efficiency while ensuring a sustainable ranching operation. The Grazing Plan distributes grazing use over upland fields and bottomland pastures better than has previously occurred. This is accomplished by shortening the grazing period, increasing stocking density, and rotating livestock among the various field-pastures within the Ranch's uplands and bottomlands. The plan will be revised every 3 to5 years incorporating amendments.

The grazing plan relies on a two-pasture deferred rest-rotation strategy. The grazing rotation strategy will provide alternate year spring growing season (May to mid-July) rest on upland forage and summer rest (late July-August) on riparian plants. Two upland pastures will be grazed, augmented by short grazing rotations through approximately a dozen smaller bottomland fields. A two-pasture rest-rotation system (grazing early one year, late the next) benefits both uplands and riparian areas by resting during the alternate year's growing seasons. Development of the pasture rotation system required maintenance on existing fences, construction of riparian fence exclosures that also served as pasture boundary fences, and creation of off-site or shared water sources for livestock as they moved from pasture to pasture.

Adaptive management will be used to monitor, evaluate, and maintain a healthy sustainable economic ranching operation that meets stream and fisheries conservation goals. Proper use of key forage species on the range units is no more than 50% of the annual plant growth. Half of the annual growth should remain on the land for plant health, watershed and wildlife values. Forage levels will be monitored and assessed by strategic placement of small-enclosed forage utilization cages.

Conservation Plan

The Conservation Plan goal is to improve fish and wildlife habitats on the Ranch, focusing on Crow Creek and its tributaries and on their riparian zones, where much of the long-term impact of cattle grazing has occurred. Impacts are greater on tributary streams (Sage, Bull, and Rock creeks), than on Crow Creek. At this point, impacts on Sage and Bull creeks have been addressed, though grazing impacts still occur on Sage Creek upstream of the Ranch. Reducing or eliminating livestock impacts on stream banks and riparian areas will allow stream banks and riparian vegetation to rebuild and stabilize.

Annual monitoring of water quality, aquatic macroinvertebrates, stream habitat features, and fish population attributes will be conducted by Formation Environmental (previously NewFields Boulder and HabiTech) out of Boulder Colorado. Results from their annual monitoring will be used to assess changes in stream habitat and quality and fish abundance and biomass and whether the habitat work on Crow Creek and its tributaries is benefiting fish populations within the Ranch.

Recreation Plan

The role of recreation on the Ranch has yet to be determined. A number of possibilities exist for public fishing and hunting opportunities, if desired. Fishing activities on the Ranch offer the opportunity for considerable public interest and good will – given the current very limited access for the public over most of the Crow Creek drainage. Fishing activities should be consistent with the Ranch's conservation and sustainability goals. This can be achieved through catch-and-release fishing using artificial lures and flies with barbless hooks. Opportunities also exist for hunting grouse and big game (deer and elk) on the larger upland ranch properties. Many western ranches allow fall hunting access on a fee basis, which can generate significant income for a western ranching operation.

Future Conservation and Restoration Opportunities on the Ranch

Future conservation work on the Ranch will focus on monitoring habitat improvements and reconnecting Sage, Bull, and Rock creeks to Crow Creek (Figure 4). The tributaries are important to cutthroat trout for spawning and for juvenile rearing. Rehabilitating and reconnecting these should increase trout numbers and biomass on the ranch.

Two conservation actions on the Ranch are our next priority. The first is to reconnect Bull Creek to Crow Creek using a series of rock-terraced pools to raise the stream up to the culvert bottom so fish passage can occur (Figure 4; left photo). The second project is to return Rock Creek to its historic stream channel on the southeast Ranch corner where it travels approximately 0.3 to-0.5 mi through wetted meadows before joining Crow Creek. Rock Creek has downcut into the wet meadow, dropping the

Figure 4. Reconnecting Bull Creek to Crow Creek will occur by using rock-terraced pools to raise the lower part of Bull Creek to reach the lower lip of the culvert under Crow Creek Road. The right photo shows downcutting of the stream in the Rock Creek pasture.

water table there, and exhibiting meter-tall eroding stream banks (Figure 4; right photo). The historic stream channel for Rock Creek is visible and a small diversion on the upper portion of the ranch should allow the stream back into its natural course. This would very quickly raise the water table in the pasture and start the process of stream rebuilding over this section with grass-stabilized streambanks already in place.

Future Opportunities – The MP Ranch as a Model

We view the Meade Peak Ranch Management Plan as a potential model for balancing the traditional western livestock ranch with fish and wildlife conservation. Too often, ecological rehabilitation of grazing impacted aquatic systems in the western United States to benefit fish and wildlife is portrayed as inconsistent or antagonistic with operation of a working livestock ranch. We believe that benefits to both the working ranch and fish and wildlife can be achieved by balancing habitat improvements with improved livestock management, particularly through use of a pasture rest-rotation strategy that increases the grazing intensity (density of stocking) but shortens the grazing duration on specific fields and pastures. If this approach proves successful, as it has elsewhere, we hope to work with other ranchers, grazing associations, and environmental groups to promote this conservation approach as a model system that can be replicated by willing participant ranch owners.

Literature Cited

Armour, C. L., D. A. Duff, and W. Elmore. 1991. The effects of livestock grazing on riparian and stream ecosystems. Fisheries 16(1):7–11.

Clary, W. P., and W. H. Kruse. 2004. Livestock grazing in riparian areas: environmental impacts, management practices and management implications. Pages 239-259 in Riparian areas of the southwestern United States, M. B. Baker et al., eds. Lewis Publishers, Boca Raton.

Idaho Fish and Game (IDFG) 2007. Management plan for conservation of Yellowstone cutthroat trout in Idaho. April 2007. 110 pp.

Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications. a review. Journal of Range Management 37:430–438.

Knapp, R. A., and K. R. Matthews. 1996. Livestock grazing, golden trout, and streams in the Golden Trout Wilderness, California: impacts and management implications. North American Journal of Fisheries Management 16:805–820.

Meyer, K. A., and J. A. Lamansky. 2004. Status of Yellowstone cutthroat trout in Idaho. in. Assessment of Native Salmonids above Hells Canyon Dam, Idaho. 2003 Annual Report for U.S. Department of Energy, Bonneville Power Administration Project Number 1998-00200, Idaho Fish and Game Report Number 04-26, Boise, ID.

McInnis, M. L., and J. McIver. 2001. Influence of off-stream supplements on streambanks of riparian pastures. Journal of Range Management. 54:648-652

NewFields Boulder and HabiTech. 2008. (Draft) Interim data report – Spring 2008: Field monitoring studies for developing a site-specific selenium criterion. Report to JR Simplot Company. October 8, 2008. 27 pp. and appendices.

Platts, W. 1991. Livestock Grazing. Pages 389-423. Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats.Special Publication (add number), American Fisheries Society, Bethesda, Maryland.

Porath, M. L., P. A. Momont, T. DelCurto, N. R. Rimbey , J. A. Tanaka, and M. McInnis. 2002. Offstream water and trace mineral salt as management strategies for improved cattle distribution. Journal of Animal Science. 80:346-356

Saunders, W. C., and K. D. Fausch. 2007. Improved grazing management increases terrestrial invertebrate inputs that feed trout in Wyoming rangeland streams. Transactions of the American Fisheries Society 136:1216-1230.

Williams, R. N., and C. D. Williams. 2006. Review of Hartman Ranch restoration opportunities and Smoky Canyon phosphate mining operations. Report to Trout Unlimited. October 30, 2006. 16 pp.

Williams, R., W. Burkhardt, and W. Platts. 2010. Meade Peak Ranch: Land and livestock grazing management plan (Crow Creek basin, Idaho). Final Report to JR Simplot Company, 30 June 2010. 66 pp.

ARE NATIONAL FISH HABITAT PARTNERSHIPS, THE NATIONAL FISH AND WILDLIFE FOUNDATION, STATE WILDLIFE ACTION PLANS AND THE LANDSCAPE CONSERVATION COOPERATIVES ENOUGH TO CONSERVE WILD NATIVE TROUT?

Robin F. Knox
Western Native Trout Initiative, 134 Union Blvd., Suite 665, Lakewood, CO 80228

ABSTRACT — Since 2006, with the advent of the National Fish Habitat Action Plan, there has been a concerted effort to address coldwater fish habitats by numerous recognized Fish Habitat Partnerships. The Western Native Trout Initiative, the Eastern Brook Trout Joint Venture, and the Driftless Area Fish Habitat Partnership have directly applied millions of dollars to trout habitat and trout populations. The National Fish and Wildlife Foundation has partnered with Trout Unlimited, the National Fish Habitat Partnerships, and federal agencies to apply more than 80 million dollars to trout management, much of that addressing both native and wild fish. The State Wildlife Actions Plans have also addressed native trout needs, but often funding for trout projects is lacking due to decreasing state budgets. The concept of Landscape Conservation Cooperatives has recently been proposed by the US Fish and Wildlife Service and may offer an opportunity for climate-change related studies on the impacts of a warming climate on native trout. The sum impact of all these actions has not been widely discussed or promoted, and the question remains – is the combined effort sufficient to turn the tide of threats currently facing wild trout?

INTRODUCTION

In 2006, The National Fish Habitat Action Plan (NFHAP) was published (AFWA, 2006), and 5 regionally-based organizations were designated as “pilot” National Fish Habitat Partnerships. The Eastern Brook Trout Joint Venture, the Driftless Area Restoration effort, The Matanuska – Sustina Basin Salmon Partnership, and the Western Native Trout Initiative (WNTI) have similar missions to serve as catalysts for the implementation of conservation or management actions, through partnerships and cooperative efforts, that result in improved native or wild trout and salmon population status, improved aquatic habitats, and improved recreational opportunities for native trout anglers.

These original Partnerships are collaborative conglomerates of state natural resource agencies, the National Fish Habitat Action Plan, the U.S. Fish and Wildlife Service, U. S. Forest Service, Bureau of Land Management, Natural Resource Conservation Service, and many tribal and public or private conservation-minded organizations.

The WNTI, as well as the other Fish Habitat Partnerships (FHPs), are non-regulatory, science-based, cooperative conservation efforts designed to speed the implementation of actions benefitting trout and their habitats. The funding that is available to these partnerships has grown from US$440,000 in 2006 to $3.35 million in 2010.

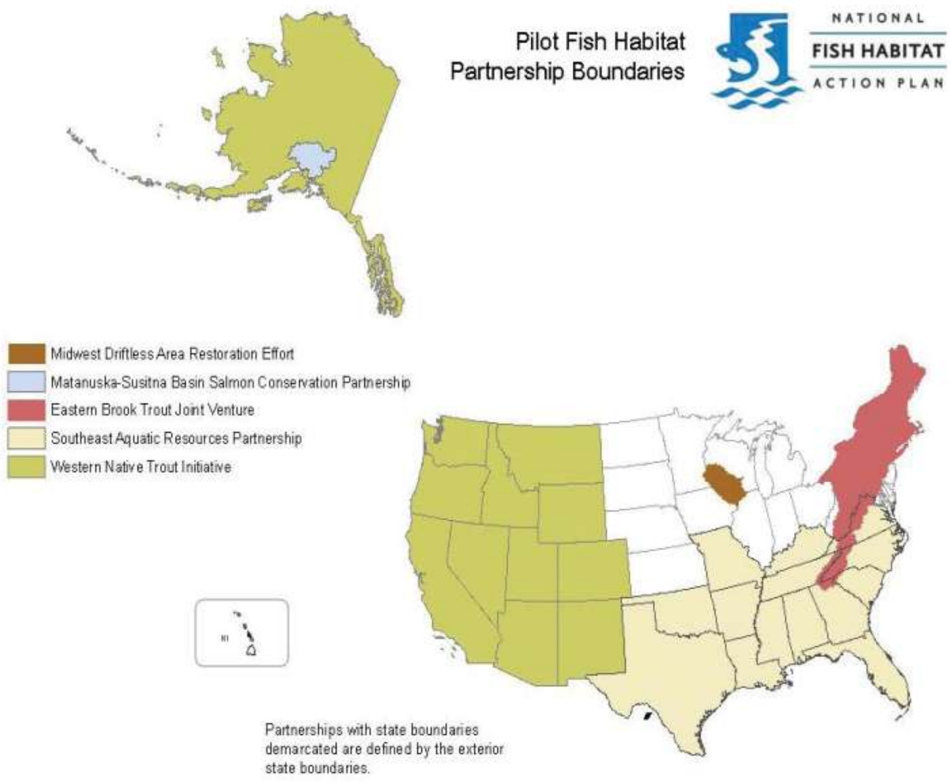


Figure 1. Map of the original NFHAP Partnerships.

The number of recognized NFHAP partnerships has grown as well, with two additional Alaska habitat partnerships, the Great Lakes Habitat Partnership, the California Fish Passage Forum, and the Reservoir Habitat Partnership all having components that address coldwater species and their habitats. However, lacking any congressionally authorized increases in funding, the level of funding directly to FHPS is likely to remain static, and the distribution to larger FHPs like WNTI and EBTJV are likely to decrease in the near future.

Cumulative Funding and Projects of the Trout-Specific NFHAP Partnerships

During the time period of 2006 – 2010, the three NFHAP Partnerships dealing with coldwater trout fisheries in the lower 48 sates, have had a remarkable record of success in matching the NFHAP funding with partner dollars.

Eastern Brook Trout Joint Venture

The Eastern Brook trout Joint Venture (EBTJV) has designated funding to 52 projects that conserve or enhance the status of brook trout. The EBTJV matched $2.6 million of NFHAP funds to $5.6 million partner match.

Driftless Area Restoration Effort

The Driftless Area Restoration Effort (DARE) has designated funding to 33 projects that conserve or enhance the status of wild trout. The DARE matched $1.47 million of NFHAP funds to $3.8 million partner match.

Western Native Trout Initiative

The Western Native Trout Initiative (WNTI) has designated funding to 60 projects that conserve or enhance the status of western native trout. The WNTI matched $2.86 million of NFHAP funds to $4.3 million partner match.

Other Key Funding Partners

The development of the National Fish Habitat Action Plan has mobilized various interests that have a desire to improve trout habitat across the United States. Two groups, Trout Unlimited (TU) and the National Fish and Wildlife Foundation (NFWF) especially have jumped on board to address critical habitat issues.

Trout Unlimited has a long-standing tradition of addressing wild and native trout habitat and conservation issues, and works on a variety of fronts to bring funding to the table for conservation projects. They work on a local level with the *Embrace A Stream* program, many state-specific or regionally-specific programs like the *Western Water Project* , and on national level with their *Bring Back the Natives Program*.

The National Fish and Wildlife Foundation also has a long history of being involved in the conservation of fish and their habitat in many areas of the country. When the National Fish Habitat Action Plan was developed, these two organizations were heavily involved. As the NFHAP has matured, they have collaborated in using the TU *Bring Back the Natives Program* as a springboard to funnel funding from a variety of sources to trout preservation, protection and enhancement projects across the country. Key partners in this effort have been the U.S. Forest Service, and the Bureau of Land Management.

In addition, the NFWF has developed several other funding categories through which they can fund trout conservation projects. These include special keystone species efforts such as the *Gila Trout recovery program*, the *Native Fish Habitat Initiative in Pacific Northwest States*, and the *Jackson Hole One-Fly Conservation Partnership.*

In the western States alone, in the 2005-2009 time frame, NFWF has matched almost $23 million in funds to 288 coldwater trout related projects with a match value of over $59 million. This is a remarkable accomplishment in an era where project funding dollars are beginning to become scarce. (*NFWF, pers. comm.)*

Landscape Conservation Cooperatives and Fish Conservation Areas

Landscape Conservation Cooperatives

In 2009, The U.S. Fish and Wildlife Service launched the Landscape Cooperation Conservation (LCC) effort. The LCC website (*http://www.fws.gov/science/shc/lcc.html)* states:

> "Landscape Conservation Cooperatives are management-science partnerships that inform integrated resource management actions addressing climate change and other stressors within and across landscapes. They will link science and conservation delivery. LCCs are true cooperatives, formed and directed by land, water, wildlife and cultural resource managers and interested public and private organizations. Federal, state, tribal, local government and non-governmental management organizations are all invited as partners in their development.
>
> LCCs will enable resource management agencies and organizations to collaborate in an integrated fashion within and across landscapes. LCCs will provide scientific and technical support to inform landscape-scale conservation using adaptive management principles. LCCs will engage in biological planning, conservation design, inventory and monitoring program design, and other types of conservation-based scientific research, planning and coordination. LCCs will play an important role in helping partners establish common goals and priorities, so they can be more efficient and effective in targeting the right science in the right places. Products developed by LCCs will inform the actions of partners and other interested parties in their delivery of on-the-ground conservation."

Although LCCS are not yet fully developed, they offer additional opportunity for the identification of actions related to climate change that could positively benefit wild trout habitats. The continued identification of actions however, does not solve the need for additional funding to implement those actions.

There is also concern that the LCC effort may be duplicative of efforts that have been completed for certain areas such as the NFHAP national fish habitat assessment and Trout Unlimited's Conservation Success Index (CSI) that focuses on key habitats and watersheds for potential conservation actions. Presently, there has been involvement by the Eastern Brook Trout Joint Venture in the Northeastern LCCs, but no involvement of WNTI in the western LCCs that encompass the geographic area of the WNTI.

The location and names of the proposed LCCs are shown in Figure 2 below.

State Wildlife Action Plans

Developed by every state and territory, the state wildlife action plans (SWAP) outline the steps that are needed to conserve wildlife and habitat before they become rarer and costlier to protect. Taken as a whole, they present a national action agenda for preventing wildlife from becoming endangered.

Often referred to as the “Teaming with Wildlife” program, Congress charged each state and territory with developing a statewide wildlife action plan to make the best use of the federal funds provided through the Wildlife Conservation and Restoration Program and the State Wildlife Grants Program. These proactive plans, known technically as “comprehensive wildlife conservation strategies,” will help conserve wildlife and vital natural areas before they become more rare and costlier to protect. As our communities grow, the wildlife action plans will help us fulfill our responsibility to conserve wildlife and the lands and waters where they live for future generations.

Unfortunately, due to their status as sport species, many of the states with native trout find it difficult to use state wildlife grant funds for wild trout conservation, unless there is a utilitarian relationship where habitat or watershed protection for a non-trout species also conserves trout habitat. The value of the SWAPs to wild trout has been mixed across the West.

Figure 2. Map of Proposed Landscape Conservation Cooperatives

CHALLENGES THAT REMAIN

Turning the Tide

Trying to turn the tide of threats poised against native and wild trout seems at times to be an indomitable task. With the continued demands for energy development, the continued potential impacts of climate warming and potential changes in the availability of water, especially in the West, all of the programs described above might seem a bit like the old Greek proverb of Sisyphus pushing that boulder uphill.

The hope is that the accomplishment of multiple projects and improving the science and base knowledge of how to best manage trout has helped to slow down, and in some case reverse declines of these iconic species.

The commonly accepted paradigm that successful management rests in the addressing of connectivity and quality of habitat in key watersheds remains intact. The pooling of strengths of many agencies and organizations remains *apropos*. The National Fish Habitat Action Plan has set a specific course of action that fits well with new efforts like the Landscape Conservation Cooperatives, State Wildlife Action Plans, recently initiated thinking about Fish Conservation Areas (Williams and Tabbert, 2006) and potential philanthropic efforts spearheaded by the National Fish and Wildlife Foundation.

Future success will be measured against how well those of us dedicated to the task of wild and native trout conservation have accomplished the following goals:

- The National Fish Habitat Conservation Act somehow makes its way through the U. S. Congress and provides a stable, but much larger base of funding for the nationwide effort to protect, preserve, and enhance trout habitat across the country.
- Combining the efforts of fish habitat Partnerships, Landscape Conservation Cooperatives, and State Wildlife Action Plans to identify and improve key watersheds and habitats for wild and native trout.
- Telling of the story of what has been accomplished, and what habitats have been protected, and what species have been improved in status despite the continuing impacts of human activities.

For further information on the National Fish Habitat Action Plan, visit:www.fishhabitat.org

For further information on the Western Native Trout Initiative, visit: www.westernnativetroutinitiative.org

For further information on the Eastern Brook Trout Joint Venture, visit:www.easternbrooktrout.org

For further information on the Driftless Area Restoration effort, visit:www.darestoration.com/

For further information on Landscape Conservation Cooperatives, visit:*www.fws.gov/science/shc/lcc.html*

For further information on the National Fish Habitat Action Plan, visit:www.fishhabitat.org

For further information on the National Fish and Wildlife Foundation trout restoration programs, visit: www.nfwf.org

REFERENCES

AFWA (Association of Fish and Wildlife Agencies). 2006. National Fish Habitat Action Plan. Washington, D.C.

USFWS (U.S. Fish and Wildlife Service). 2009. Strategic Habitat Conservation – LCC, Description. Washington, DC.

Williams, R. N., and R. Tabbert. 2005. Protecting resident native trout: a case for a national refuge system. Flyfisher, Winter Issue. Pages 30-33.

WHY RESTORE? CULTURAL AND ECOLOGICAL REASONS FOR NATIVE TROUT RESTORATION

Samuel Snyder, PhD
John H. Daniels Fellow, National Sporting Library, Middleburg, VA, Assistant Director, Renewable Resources Coalition, Anchorage, Alaska, snyderaway@gmail.com

ABSTRACT — Using the Rio Grande cutthroat trout *Oncorhynchus clarkii virginalis* as an example, this paper builds on extensive field work in New Mexico to explore the cultural elements of native trout restoration. In doing so, I argue that understanding the cultural justifications for native species restoration is essential for building future support for similar initiatives elsewhere as well as negotiating environmental conflict. I begin by exploring the reasons for involvement in native trout restoration, paying special attention to the role of anglers in restoration initiatives. Second, I examine the potential conflicts that might arise amidst the restoration of native trout, including resistance from other anglers who might prefer introduced brown trout *Salmo trutta*, for example. Finally, drawing on insights from the larger field of ecological restoration, I underscore potential cultural and community benefits for native species restoration, particularly the encouragement of place-based collaboration integral for the protection of wild spaces and amidst other threats to trout ecosystems.

INTRODUCTION

In early 2000, representatives from New Mexico Trout, an organization "dedicated to the preservation and enhancement of trout fishing in New Mexico's waters through restoration of riparian habitats and through the education of the public about trout fishing and the value of trout habitats" , contacted the Quivira Coalition, also a New Mexico-based grassroots organization with the mission of building "resilience by fostering ecological, economic and social health on western landscapes through education, collaboration, and progressive public and private land stewardship." These two groups contained outwardly different constituents – one fly fishers, the other ranchers. Yet, their collaboration converged in native trout restoration on Comanche Creek in northeastern New Mexico's Valle Vidal. Within several summers, their initiatives grew to include volunteers from Trout Unlimited, the Boy Scouts, Student Conservation Association, Amigos Bravos, and more. By the summer of 2005, their restoration work was well underway, while other threats to the region emerged - mainly oil and gas exploration. The restoration work in the Valle Vidal not only returned native trout to home waters, but also would prove essential for building bedrock community that was vital for protecting the Valle Vidal from threats posed by oil and gas exploration. Participating in and researching this story as a sociologist concerned with grassroots environmental collaboration and environmental conflict resolution and as a philosopher trained in environmental ethics, I came to realize that the restoration of the Valle Vidal's Comanche Creek had as much to do with restoring culture as it did restoring nature.

Recreational anglers have long played a pivotal role in American fisheries management and conservation. The economics of tourism, aesthetic valuations of trout, and personal preferences of sport fishing have often dictated hatchery management and fish stocking programs. In America and around the globe, the introduction of trout has altered cold-water ecosystems. Today, however, small pockets of angler-conservationists are banding together to move beyond "wild trout" in support of "native trout." While still controversial in some circles, anglers, conservationists, and other stakeholders are working together to restore native trout, such as the Rio Grande cutthroat trout Oncorhynchus clarkii virginalis, thanks to the combination of conservation biology and the leadership of grassroots initiatives. Beyond ecological justifications, the restoration of native trout carries with it significant cultural and political implications.

Using the Rio Grande cutthroat trout as an example, this paper builds on extensive field work in New Mexico to explore the cultural elements of native trout restoration. In doing so, I argue that understanding the cultural justifications for native species restoration is essential for building future support for similar initiatives elsewhere as well as negotiating environmental conflict. I begin by exploring the reasons for involvement in native trout restoration, paying special attention to the role of anglers in restoration initiatives. Second, I examine the potential conflicts that might arise amidst the restoration of native trout, including resistance from other anglers who might prefer introduced brown trout *Salmo trutta*, for example. Finally, drawing on insights from the larger field of ecological restoration, I underscore potential cultural and community benefits for native species restoration, particularly the encouragement of place-based collaboration integral for the protection of wild spaces and amidst other threats to trout ecosystems.

Becoming Native to Place

While he might not be well known among fly fishers, trout enthusiasts, or fisheries managers, Wes Jackson is a scientist and philosopher from whom I believe we could all learn something as we wrestle with the conservation, preservation, and restoration of trout and their coldwater habitats. Jackson is a biologist, plant geneticist, philosopher, and founder of the Land Institute in Salina, Kansas, where he works on perennial grasses and plants searching for ways to combat soil erosion, pesticide use, and other prominent issues facing American agriculture. What interests me, as a fly fisher, environmental educator, and conservationist, is not necessarily his work on perennial grasses, but his philosophical approach to place.

Jackson (1994) mused that "the majority of solutions to both global and local problems must take place at the level of the expanded tribe, what civilization calls community. In effect," he explains, "we will be required to become native to our little places if we are to become native to this place, this continent. . .". Jackson's insistence on becoming native to place is one voiced in a variety of ways by philosophers, writers, and activists who identify with bioregionalism and the importance of place. Many of these writers are or have been fly fishers and advocates for trout and salmon conservation – Roderick Haig-Brown, among others, comes immediately to mind. But what does it mean exactly to "become native to place"? Understanding how humans might become native to their places is an interesting issue to ponder particularly in the midst of the various efforts to not only preserve but to restore populations of native trout to their native waters.

Fly Fishing and Mixing Trout

Throughout American environmental history, anglers (notably fly fishers) have been instrumental in messing with, managing, conserving, and restoring coldwater ecosystems. Their engagement in the fate of waters stems from experiential values of experiencing nature. Anglers in the late 1800s, such as Thaddeus Norris or Seth Green, often romantically reflected on the fly fishing experience, tied that experience to a reflection on deteriorating waters, while advocating some form of conservation. Those sentiments, I have found, remain true today as they did the moment early American angler-pisciculturists began, as Aldo Leopold (1918) wrote, "mixing trout in western waters" or any waters for that matter. In doing so, however, their work (despite good intentions) was the exact opposite of Jackson's discussions of being native to place.

In a story, we are all too familiar with, our management approaches have since the 1880s centered around faulty assumptions on genetics, geography, and were driven by aesthetics, angler-centered preferences. As Paul Schullery (1999) wrote, "as we have introduced nonnative fish not only to fishless waters but also to waters containing native fish, we have lowered a kind of ecological eggbeater into some glorious native ecosystems, resulting in changes that, though they may have been wonderful for fishermen, were disastrous for these little worlds that had been cranking along just fine without our help since the last ice age". Yet, over time, he mused, "Our standards and values have changed and our understanding of wild ecosystems has evolved so that we have higher expectations when we insist on wild trout in our streams".

By the 1970s, any trout would not do. Management goals and angler desires began paying attention to wild trout. This reality evidenced by the first Wild Trout Symposium hosted in 1974. Today, we have higher expectations. We hope for wild trout in our streams, and moreover, we are increasingly working to restore native trout in their native streams, and not just because anglers want to catch them, but we increasingly understand that they belong there (Williams 2005; Schullery 2006).

Restoring Rio Grande Cutthroat Trout

As many of you understand, the Rio Grande cutthroat trout – the first trout written about in the "new world" by Europeans - has over the 500 years since that fateful encounter with the expedition of Francisco Vasquez de Coronado declined to a state where they currently occupy roughly 7% of their native range (Trotter 1987; Stefferud 1988). These numbers are low, but stable. Over time, anglers have come to appreciate those dwindling numbers and decided to work to restore small pockets of cutthroat trout. In my field-based research, I interviewed anglers about their desires and motivations to restore native trout.

Anyone familiar with the deep (and sometimes over the top) literary history of fly fishing knows that an increasing number of fly fishers believe that angling for native trout, on solitary, isolated streams represents the archetypal fly-fishing experience. Looking back at a history of trout (mis)management, some compare the quest for native trout with the more profane excursions of catching nonnative, hatchery trout on a crowded stream. David James Duncan (2001) elevated native fish when he explained that the opposite of native is hatchery. Of course, as we know, wild trout sit somewhere in between the hatchery and the native, yet even John Gierach (1989) ranted that:

> I don't care much for hatchery trout. They're better than no trout at all, but otherwise they're inferior in every way to their wild relatives . . . Hatchery fish are, well. . . they're from a hatchery; they don't seem to belong in the stream, they're often the wrong species (rainbows where cutthroats should live, for example), most are pale and sickly looking when compared to wild fish and, having been raised on Purina Trout Chow, they aren't very good to eat.

Of course, we certainly value wild trout. The turn to wild trout by the 1970s, however, represented more than a preference for trout by anglers and was hinged upon more thorough understandings of the ecologies of fishes and their watersheds. The concern with native trout is an extension of that perspective. Take the Anglers Life List Website (http://anglerslifelist.com) as proof of a reverence for native trout like the Rio Grande cutthroat trout. This is a website where anglers can share photos, discuss in forums, and generally celebrate the experiences of catching native trout in their native waters.

Aside from a solitary and secluded fly fishing experience though, concern for native trout is the result of more that angling desires and is rooted in the knowledge gained through the angling experience. Anglers interviewed over the course of several years of fieldwork in contexts of native cutthroat trout and brook trout *Salvelinus fontinalis* restoration projects around the country revealed a common sentiment. I have heard that anglers start with fly fishing, then they learn about bugs, or flies, of fly fishing, which leads to rudimentary streamside ecology. This expanded vision, anglers often reflected, produces a picture of the whole upon which native trout depend. That same vision also revealed that streams remain on the decline, habitats suffer from neglect, overfishing, logging, or pollution. There is a combined knowledge and concern that then gives rise to engaged activism with their local Trout Unlimited chapter and the hands on work of native trout, stream restoration. In these moments, anglers have moved toward valuing the native trout not because they want to catch them, but because its place in the ecosystem means something both historically and tangibly in the present. Anglers like knowing those fish are there; they learn to see the cutthroat trout, for example, in the context of the entire biotic system of life, rather than merely a game species.

Seeing the system whole, anglers can begin to understand the necessity of maintaining all of the parts of the biotic system. Echoing, Aldo Leopold (1949) (also an avid fly angler) who wrote that "a thing is good when it tends to preserve the integrity, stability, and beauty of the biotic system, it is wrong when it tends otherwise" , and understanding that if the parts of the biotic system are not intact, then restoration is a necessary course of action. However, restoration is highly controversial.

Debating Restoration

As many of you know and understand, ecological restoration represents the attempt to make landscapes whole again. Restoration specialist Dan Daggett (2005) echoed Wes Jackson when he noted that restoration is not only the attempt to restore native species, but it "offers a way to become native once again". The human can potentially become native to place through restoration, because restoration demands "understanding the historical conditions that led to present conditions" (Higgs 2003). In the case of native trout like the Rio Grande cutthroat trout, for example, restoration entails not only restoring trout to the native waters, but restoring the watersheds themselves, all the while understanding why those waters are crucial for their survival and all of the historical threats that have put both fish and rivers in jeopardy. In New Mexico, for example, this meant understanding the role of cattle ranching as much as if not more than deforestation upon the high desert rivers. This is a reality that has not only led to anglers restoring rivers, but anglers working with ranchers to restore rivers and find more sustainable means of managing livestock and the ranching community so vital to the cultural identity of New Mexico.

Often collisions between ranchers and trout enthusiasts have given rise to conflict. However, across the American west, the two stakeholder groups are finding ways to work together to mutually restore waters while encouraging their respective cultures, professions, and passions. If restoration is not being debated by these groups, however, does not mean that it is not controversial. In the realms of philosophy, some environmental ethicists question the tenants of restoration.

Two of the most vocal opponents are Robert Elliot (1997) and Eric Katz (1996; 2000), who argued that restored systems are like lies that fake nature. Despite best intentions, they both believe that a restored landscape, watershed or species can never have the same value as the original system. More seriously, Elliot called restoration a lie because it might be too easily co-opted by those wishing to first degrade an ecosystem. His caution is one we must take seriously in the face of oil, gas, or mineral exploration, where remediation is touted as the end result of a development project. In the end, however, remediation is usually far from restoration and in many cases, restoration or remediation is hardly a real option given the scope of the project proposed.

Even in those contexts where a system is already degraded from years of misuse or neglect, Katz (1996) wrote that "the practice of ecological restoration can only represent a misguided faith in the hegemony and infallibility of the human power to control the world". In other words: hubris.

Yet, restoration need not equal control of nature. New Mexico restoration specialist Bill Zeedyk teaches a method of stream restoration called "induced meandering," which requires humans to build small barriers into a stream while allowing nature to take its course and reshape or rebuilt bank structure over time. Induced meandering provides a nice philosophical model for avoiding hubris in any restoration project because it takes the impetus off of humans and places it on the processes of nature. Or, it encourages a sense of collaboration between the two. Yet, philosophical debates are hardly the major hurdles, on the ground conflicts tend more frequently to stall collaborative restoration.

For example, the environmental advocacy group Wilderness Watch warns that restorations of this sort are about sport, not a concern for biodiversity. The point, they argued, "is to remove stocked trout and replace them with listed trout in an effort to boost the population to a level that will allow delisting and resumed sport fishing of these species" (Williams 2005). While this might be true for some, it hardly holds water in all cases. Anglers, without getting too proud of themselves, must remember have been a valuable fight in the push for species listing, all the while recognizing that if the trout is listed as endangered, then sporting opportunities might decline.

Endangered species listing aside, angler enjoyment of catching native trout in their native and isolated location is not sufficient reason for restoration,although the lure of fishing for native trout is a powerful way to develop advocates for those ecosystems. Yet, according to author Ted Williams (2005), anglers should "defend native fish not because they are fun to catch or good to eat or beautiful, not because they are anything, but because they are". Others have echoed his point most notably Paul Schullery (2004), who has addressed this audience more on more than a few occasions. Despite these holistic proclamations, however, there is also sufficient debate within the angling community on the matter.

Some anglers do not see the need to mess up one perfectly good trout stream in the name of another trout stream. Within the history of fly fishing anglers have debated the merits of different trout, from the newly arrived brown trout in the early 1880s, to the transplanted and globalized rainbow trout, and not of course the merits of native cutthroat trout or brook trout. Too often anglers preferences have driven the course of our management decisions, which is why we are to some degree in the predicament we are in today. Using history to advocate for restoration, however, Bill Schudlich of New Mexico Council of Trout Unlimited (TU), explained that cutthroat trout need restoration because "the native fish are as much a part of our natural heritage as the Carlsbad Caverns and the Rio Grande. If we don't restore these fish, something unique and special about New Mexico will be lost" (2006). Yet, even if one agrees on the end result, many fear the process. The most contentious of all issues revolves around not only the methods for removing nonnative species, but the act of removing them altogether.

The use of piscicides Antimycin A and rotenone are debated by many, causing controversy over the perception that anglers are "poisoning rivers" for the sake of restoring game fish. Yet, most organizations involved in this work recognize that electrofishing alone is not an efficient means of removing nonnative and invasive species.

Whether using chemicals or electrofishing techniques, restoration projects always entail some level of violence (i.e., killing fish), and that makes some more than uneasy. However, fisheries biologist or restoration ecologists will remind you that acts of introducing nonnative fish were equally violent in their own right. Repairing the damage from one form of violence unfortunately necessitates a bit more violence. Reflecting on this double bind, restoration ecologist William Jordan III (2003) likened the goals of restoration to the repayment of a debt. "Everything we take from nature," he reflected, "sometimes by persuasion or collaboration, sometimes by outright theft. Either way, the debt we incur is, or ought to be, a constant concern. For many, restoration is an attractive idea because it offers a way of repaying that debt" . This repayment involves difficult decisions and actions, but as Sean Farrell of the U. S. Fish and Wildlife Service explained in an interview, "you cope with the pain by realizing that in several months to years, through proper attention to restoration, the river should be healthier than it was without those native species." Restoration, therefore, provides outlets for making ecosystems whole again. Yet, ecological restoration, in many ways is more than about restoring ecosystems, but about restoring community, building coalitions, or as Jackson said, "becoming native to place."

Restoring Community – Ecological Citizenship

In *Bright Waters, Bright Fish* (1980), Roderick Haig-Brown connected fly fishing and other forms of outdoor recreation to the development of a scientific understanding of nature and an "ethic of land, air, and water". Like fly fishing, restoration ecology can and should foster the development of a scientific appreciation of nature, as well as a deepened ethic of respect for nature. Environmental ethicist J. Baird Callicott (2002) argues that restoration should teach about the spatial and temporal fit of a species to an ecosystem. Fit, he insists, not necessarily history, is a crucial parameter; a restoration makes sense if the species in question still fits in the ecosystem. In the case of Rio Grande cutthroat trout, assuming all predators aside, they still fit in the cold headwaters of New Mexico streams, and the same goes for other native trout around North America. The focus on fit recalls Jackson's discussions of "native," as he believes humans have forgotten how to work with nature rather than work against it, as we do nowadays. In other words, humans need to learn how to fit into their bioregions.

If fishing, as many fly fishers argue, can teach us about fish or streamside ecology, and if our contemplation of native fish might reveal what is wrong with an ecosystem or watershed, then restoration projects offer ways for us to actively engage those issues. Indeed, many are making this connection, as is evidenced not only by statements and policies from TU, Federation of Fly Fishers, and countless grassroots groups like New Mexico Trout, but also through their endeavors in projects like TU's Bring Back the Natives or the Western Water Project. Restoration, unlike fishing, however, is a highly collaborative enterprise. If fly fishing is celebrated for the solitude it brings the angler, restoration can bring anglers and other concerned stakeholders together to work for the common good of the community and bioregion. Becoming "native" to place has as much to do with learning how to work with and for nature as it does negotiating how to work in human communities.

Restoration, can lead to what Andrew Light (2002) has referred to as 'ecological citizenship,"

> The goal of an ecological citizenship is to bring together the interests of a human community to be fair and open and conducive to allowing each member of a community to pursue his or her own private interests while also tempering these pursuits with attention to the environment. A strengthened relationship with nature is to be found in forming open-ended organizational bonds that entail specific moral, and possibly legal, responsibilities to create for the nature around one's community and respect the environmental connections between communities.

According to Light (2007), "one of the more interesting things about ecological restorations is that they are amenable to public participation". State or federal agencies often rely on volunteers from TU, local watershed groups, student conservation clubs, and more. The work of restoration itself always demands group involvement, from building fences, to moving rocks, or even providing and sharing meals together at the end of the day. All of this collectively builds strong community, which is vital for the long-term protection of an entire watershed. Restoration provides one of many outlets for getting to know and celebrate one's local bioregion while potentially building sustainable communities. This includes getting to know the local cultures, history, art or food as much as it does the local biodiversity. Light (2002, 2007) is correct, and nowhere is this reality more evident than in what would become the Coalition for the Valle Vidal.

Concluding on the Coalition for the Valle Vidal

If restoration of native cutthroat trout began in 2000, the collaborative work would become more meaningful in 2004 and 2005 when El Paso Oil Company proposed to drill in the Valle Vidal region. By early 2005, David Stalling, fly fisher, and Western Coordinator for TU called for the protection of what he called one of the most "spiritually sacred areas in the southwest" – New Mexico's Valle Vidal (literally Valley of Life). Stalling's call came in response to proposals by the El Paso Oil Company and the United States Government to explore this 90,000-acre paradise for oil. While his call went out to fly fishers and trout lovers, thousands of others responded each claiming their own reasons for protecting this sacred land – from native trout, ranching history, to migrating elk. On December 13, 2006, President George W. Bush did the unthinkable by signing a bill into law protecting the Valle Vidal from oil exploration.

While interviewing Jim O'Donnell, the director of the Coalition for the Valle Vidal, he repeatedly explained to me that he viewed early work to restore native cutthroat to the region as vital for creating what journalist Rebecca Clarren (2006) playfully termed the "coalition that could". What happened in New Mexico, and is happening elsewhere, is an example of what political scientist Edward P. Weber (2003) calls "coalitions of the unalike," which are emerging across the American landscape to forge unique manifestations of grassroots, place-based democratic alliances in defense of community and the environment.

Restoration ecology paved the way for groups of the unalike - ranchers, anglers, traditional "greenies", and more conservative conservatives, Anglos, Hispanics, and native communities - to work together to protect what many saw as a special place from potential threats of oil and gas exploration. In many ways, these movements represent bioregionalism-in-action, where place-based engagement is necessary for the creation of sustainable communities. Collaboration of this sort relies heavily on local knowledge and participation - "the individual and collective expertise of those community members most familiar" with the particular ecosystems in question (Weber 2000). Further, community entails those who reside in or have a stake in a particular place, such as an ecologically defined space, watershed or region. As Daniel Kemmis (1990) noted, "place" becomes the catalyst for community restoration, protection, and ideally in the long run sustainable management.

In these initiatives, practices in nature, such as fishing or restoration itself, provide innovative new outlets or entryway activities for individuals and groups to rethink what it might mean to understand, value, and engage both nature and community. In short, through trout restoration, anglers, fisheries managers, and trout conservationists can create opportunities to do so much more than simply restore native trout, they can facilitate the emergence of coalitions that are a bit closer to Jackson's idea of "native to place."

Acknowledgements

I am grateful to those groups who aided in my research – Trout Unlimited, New Mexico Trout, Amigos Bravos, The Quivira Coalition as well as agencies such as United States Fish and Wildlife Service and United States National Forest Service. I am grateful for all of the anglers and conservationists for leading me into their watersheds and doing the hard work of restoring native trout and their ecosystems. A version of this paper was published in American Angler Summer (2009).

References

Callicott, J. B. 2002. Choosing appropriate temporal and spatial scales for ecological restoration. Bioscience 27 (4): 409-420.

Clarren, Rebecca. 2006. “The Coalition that Could.” *Orion*. November/December: 66-67.

Daggett, D. 2005. Gardeners of Eden. Thatcher Charitable Trust Press, Santa Barbara, California.

Elliot, R. 1997. Faking Nature: The Ethics of Ecological Restoration. Routledge Press, New York, New York.

Ferrell, S. Interview with Author. 17 February 2006.

Gierach, J. 1989. Fly Fishing Small Streams. Stackpole Books, Harrisburg, Pennsylvania.

Haig-Brown, R. 1980. Bright Waters, Bright Fish: An Examination of Angling in Canada. Douglas and McIntyre, Vancouver, British Columbia.

Higgs, E. 1996. The politics of ecological restoration. Environmental Ethics 18: 227 – 248.

2001. Nature By Design: People, Natural Processes, and Ecological Restoration. MIT Press, Cambridge, Massachusetts.

Jackson, W. 1994. Becoming Native to This Place. University Press of Kentucky, Lexington.

Jordan, W. 2003. The Sunflower Forest: Ecological Restoration and the New Communion With Nature. University of California Press, Berkeley.

Katz, E. 1992. The Big Lie: The Human Restoration of Nature. Philosophy and Technology 12: 231 – 234.

Katz, E. 1996. The Problem of Ecological Restoration. Environmental Ethics 18: 222 – 224.

.

Kemmis, Daniel. 1990. Community and the Politics of Place. University of Oklahoma Press, Norman.

Leopold, Aldo. 1918. Mixing trout in Western waters. Transactions of the American Fisheries Society

47: 101-102.

Leopold, Aldo. 1949. A Sand County Almanac: And Sketches from Here and There. Oxford University Press, New York, New York.

Light, A. 2002. Restoring Ecological Citizenship. Pages 153 – 172 in B.A. Minteer and B. P. Taylor, editors. Democracy and the Claims of Nature: Critical Perspectives for a New Century. Rowman and Littlefield, Lanham.

Light, A. 2007. Restorative relationships: from artifacts to ‘natural’ systems. Pages 95 – 116 in R. France, editor. Healing Natures, Repairing Relationships: Landscape Architecture and the Restoration of Ecological Spaces. Green Frigate Books, Sheffield: 95 – 116.

Schudlich, B. 2006. Trout Unlimited $460,000 to restore New Mexico’s native fish. Press Release, 11 July.

Schullery, P. 1999. Royal Coachman: The Lore and Legends of Fly-Fishing. Simon & Schuster, New York, New York.

Schullery, P. 2004. Cowboy Trout: Western Fly Fishing as if it Matters. Montana Historical Society Press, Helena.

Stefferud, Jerome. 1988. Rio Grande Cutthroat Management in New Mexico. Pages 90-92 in R. E. Gresswell, editor. Status and Management of Interior Stocks of Cutthroat Trout. American Fisheries Society, Symposium 4., Bethesda, Maryland.

Trotter, Patrick. 1987. Cutthroat: Native Trout of the West. University of Colorado Press, Boulder.

Weber, Edward P. 2000. A New Vanguard for the Environment: Grass-Roots Ecosystem Management as a New Environmental Movement. Society and Natural Resources 13: 237-159.

Weber, Edward P. 2005. Bringing Society Back In: Grassroots Ecosystem Management, Accountability, and Sustainable Communities. MIT Press, Cambridge, Massachusetts.

Williams, T. 2005. Environmentalist vs. Native Trout. Fly Rod & Reel April: 26

Poster Presentations

EASTERN BROOK TROUT JOINT VENTURE

Doug Besler

Mountain Region Fishery Supervisor, North Carolina Wildlife Resources Commission, 645 Fish Hatchery Road, Marion, NC 28752, 828-659-8684 (ext. 221), doug.besler@ncwildlife.org

Brook trout *Salvelinus fontinalis* are a recreationally and culturally important species, regional icon, and indicator of high water quality; however, populations are declining across their historic eastern United States range (Maine to Georgia). The Eastern Brook Trout Joint Venture (EBTJV) is a partnership of state and federal agencies, nongovernmental organizations, and academic institutions. This collaborative approach to brook trout management is justified because (1) brook trout are declining across their entire eastern range; (2) causes for these declines are similar; (3) an integrated approach would be cost effective; and, (4) watersheds of concern span state borders and state and federal jurisdictions. In 2005, the EBTJV completed a range-wide assessment of brook trout populations throughout their native eastern United States range. Intact stream populations of brook trout, where wild brook trout occupy >90% of historical habitat, exist in only 5% of the watersheds assessed. The EBTJV partners agreed that a broad-scale, range-wide conservation strategy is necessary to stop brook trout declines, improve technology transfer, and effectively prioritize funds and projects to restore this important species. The Conservation Strategy is a goal-oriented, science-based, action plan that explicitly states EBTJV principal goals, presents guidance for decision-making, and provides methods for evaluating success. Findings from the range-wide status and threats assessment serve as the foundation for the development of the vision, goals, objectives, strategies, procedures, and guidelines contained within the EBTJV Conservation Strategy. The EBTJV believes this structure will result in a focused, technically credible, publicly accountable program linking EBTJV projects to specific objectives so that funding will be effectively used.

The vision of the EBTJV is to ensure "healthy, fishable brook trout populations throughout their historic eastern United States range." The principal goals of the EBTJV are (1) conserve, enhance and restore brook trout populations that have been impacted by habitat modification, or other threats and disturbances; (2) encourage partnerships among management agencies and stakeholders to seek solutions to issues such as regional environmental and ecological threats; (3) develop and implement outreach and educational programs to ensure public awareness of the challenges that face brook trout populations; and (4) develop support for implementation of programs that perpetuate and restore brook trout throughout their historic range.

INCORPORATING UNCERTAINTY INTO A WATERSHED-LEVEL ESTIMATE OF WESTSLOPE CUTTHROAT TROUT FOR THE CASTLE RIVER, SOUTHWESTERN ALBERTA, CANADA

Jason Blackburn

Fisheries Biologist, Alberta Conservation Association, 1609 - 3 Ave South, Lethbridge, Alberta, Canada, T1J 0L1, Jason.Blackburn@ab-conservation.com, Phone: 403 381-5335 Mobile: 403 894-5335. Website: www.ab-conservation.com

To avoid biases associated with site-level "index" abundance estimates, we used stratified random sampling methods at the watershed scale to systematically assess the abundance and density of threatened westslope cutthroat trout in the Castle River watershed of southwestern Alberta in 2009. We divided the Castle River watershed into five strata, from headwater tributaries to the main-stem river channel based on stream order and mean wetted width. Site-level cutthroat trout density and abundance data were collected from 93 sites using backpack, tote-barge, and raft electrofishing. We used beta distributions of simple capture probabilities (by sampling method) from capture-mark-recapture estimates to correct for single pass catches. Using corrected bootstrapped capture data for all ($\geq$70 mm fork length) westslope cutthroat trout, and those of legal-harvest size (>285 mm fork length), we extrapolated abundance and density estimates to the watershed scale. We compared estimated fish abundance and densities calculated by linear stream distance (fish/km) versus area (fish/m^2), for differences in estimates. Additionally, we compared westslope cutthroat trout abundance between watersheds that were stratified identically but incorporated different levels of uncertainty. At the watershed scale, estimated total population size of cutthroat trout by linear distance and area methods were within 5% of each other at 112,484 (90%CI = 70,728–175,775) and 106,997 (90%CI = 64,268–171,906) fish; and 4,091 (90%CI = 2,137–7,062) and 3,743 (90%CI = 2,015–6,411) for legal-harvest-sized fish. Discrepancies between estimates occurred at the stratum level from proportional contribution of strata to overall watershed abundance. Compared to area methods, linear calculations appeared to underestimate smaller strata and overestimate larger strata. Comparison of estimates between identically stratified watersheds revealed the Castle River watershed has approximately one-third the cutthroat trout abundance of the upper Oldman River watershed, and incorporating uncertainty in capture efficiency and wetted width into the Castle watershed estimate reduced the precision of our estimates 15%.

WILD TROUT SCAVENGER HUNT: BUILDING SUPPORT FOR OUR WILD AND NATIVE SPECIES

Norm Crisp

Stream Side Adventures, streamsideff@yahoo.com

Wild and native species can best be protected if they have a dedicated constituency to champion their cause. Resource managers at all levels and nongovernmental organizations (NGO) such as Trout Unlimited and the Federation of Fly Fishers are now providing the majority of the support. While the trout fishing public is providing indirect support through their membership in the conservation organizations, a more direct and active support would enhance wild trout management efforts.

Fisheries management agencies have established programs such as the “Master Angler” programs to recognize trout anglers who have caught large fish. The California’s Heritage Trout Challenge, Wyoming’s CutSlam, and the Federation of Fly Fisher’s CutCatch award recognize anglers for catching native species, but no management agency or NGO recognizes anglers for catching wild trout.

Each management agency with jurisdiction over wild species, native or not, as well as outfitters and manufactures can and should develop recognition programs and awards for anglers that successfully pursue wild species. Such programs would increase the excitement and benefits of fishing for wild trout and build their constituency. This program will provide a conceptual framework for how these programs can be developed, identify potential impediments, and suggest ways to begin implementation of these constituency building programs.

THE HENRY’S FORK CALDERA PROJECT – WILD TROUT RESEARCH AND RESTORATION

Jim DeRito, Steve Trafton, and Anne Marie Emery Miller

Henry’s Fork Foundation, PO Box 550, Ashton, Idaho 83420, jderito@henrysfork.org

The Henry’s Fork of the Snake River contains world renowned wild rainbow trout *Oncorhynchus mykiss* fisheries in the Caldera Section of the river. These fisheries have been managed for wild trout since 1978 and have had section-wide catch-and-release and artificial lure with single barbless hooks only regulations since 1987. Furthermore, these fisheries have been the subject of over 20 years of research and intensive management. Despite these efforts, anglers say that the fishery within Harriman State Park has fewer fish, diminished aquatic insect hatches, and degraded habitat as compared to past time periods. Many anglers have even stated that the solution for this apparent decline is to resume fish stocking. These issues were discussed, in part, by a panel during Wild Trout VIII and were the subject of a presentation during Wild Trout IX. In 2008, the Henry’s Fork Foundation began the Caldera Project to address anglers’ concerns. Initial project efforts focused on an assessment of the situation, including an angler attitude survey of the Henry’s Fork in Harriman State Park, a comprehensive review of scientific information, and outreach to anglers. Based on assessment results, additional efforts have been focused on addressing the limiting factor for this fish population – that of improving young-of-the-year rainbow trout survival. Furthermore, a habitat restoration process for Harriman State Park has begun. This process has included contemporary assessments of aquatic macrophytes, sediment, and geomorphology and comparisons to past conditions where possible. Habitat restoration alternatives are being developed by an interdisciplinary technical team with a focus on improving aquatic habitat for fish and macroinvertebrates in Harriman State Park and adjacent reaches of the river.

EVALUATION OF BROOK TROUT ENHANCEMENT REGULATIONS IN NORTHCENTRAL PENNSYLVANIA

Jason Detar[1], Dave Kristine[1], Tom Greene[1], and Tyler Wagner[2]

[1]Pennsylvania Fish and Boat Commission, 450 Robinson Lane, Bellefonte, PA 16823

[2]U.S. Geological Survey, Pennsylvania Cooperative Fish & Wildlife Research Unit, Pennsylvania State University, 402 Forest Resources Building, University Park, PA 16802

In 2004, the Pennsylvania Fish and Boat Commission adopted Wild Brook Trout Enhancement regulations on select waters to evaluate the effectiveness of catch-and-release regulations on improving the size structure of brook trout *Salvelinus fontinalis* populations. Under these regulations, angling is permitted on a year-round basis with no tackle restrictions, and no brook trout may be harvested at any time. Statewide regulations (178 mm; 7 inch minimum length limit and 5 trout/day creel limit from the opening day of trout season through Labor Day, and no harvest for remainder of year) apply to other trout species. Brook trout populations were sampled in six treatment and two control streams using backpack electrofishing gear for at least 1 year prior to regulation implementation and periodically over at least a 5-year period post-implementation. We analyzed electrofishing catch data to evaluate if the abundance of large brook trout ($\geq$ 178 mm; 7 inches) increased as a result of the regulations. Preliminary results 5 years into the regulation evaluation suggest that abundance of large brook trout has varied considerably over time with no strong increasing trend in the treatment or control groups. Final program evaluation is scheduled for 2011.

IMPLICATIONS OF DIVERSION DAM ON MOVEMENT PATTERNS OF BROWN TROUT IN THE ENCAMPMENT RIVER, WYOMING.

Steve Gale

Wyoming Game and Fish Department, 528 South Adams, Laramie, WY, 82070, 307-745-5180 ext. 239 (W), 307-745-8720 (F), Steve.Gale@wgf.state.wy.us

The Encampment River is a major tributary to the North Platte River and is believed to be important to the long-term persistence of trout populations in the Upper North Platte watershed. Our goal is to manage for tributary conditions that will improve trout movements, spawning, and recruitment to the North Platte River. Since the early 1900s, an irrigation diversion dam on the Encampment River located about a 0.5 mi upstream from the confluence with the North Platte River has most likely precluded spawning trout from moving upstream. A fish ladder was installed in 1985, but high spring flows in 1986 washed out the fish ladder and threatened the integrity of the diversion dam. Trout populations could benefit in the Encampment and North Platte rivers if passage was again provided. It was necessary to collect information on potential benefits before pursuing such a major fish passage project. A combination of Floy tags (N = 249 in 2007) and radio transmitters (N = 32 in 2008) was used to document movements of brown trout *Salmo trutta* collected below the diversion dam and released upstream of the dam. The final fates of radio-tagged fish were determined along with upstream and downstream migration barriers and spawning timing and locations. Spawning occurred from late September to late October and most occurred from the diversion dam to 8 river miles upstream. No other upstream migration barriers were documented, but 41% of fish encountering the diversion dam while migrating downstream were entrained in the irrigation canal. The distance between the farthest upstream relocation and downstream relocation was over 85 river miles. Providing fish passage around the diversion dam would help restore the fluvial life history of salmonids in the Encampment River watershed and enhance trout populations in the Upper North Platte River watershed.

A LARGE-SCALE ASSESSMENT OF WILD BROOK TROUT POPULATIONS AND HABITAT IN MAINE

Merry Gallagher[1], Forrest Bonney[2], and Philip Wick[3]

[1]Fisheries Research Biologist, Fisheries Research Section, Maine Department of Inland Fisheries & Wildlife, Bangor, ME

[2]Regional Fisheries Biologist (*retired*), Rangeley Region Headquarters, Maine Department of Inland Fisheries & Wildlife, Strong, ME

[3]Fisheries Research Specialist, Fisheries Research Section, Maine Department of Inland Fisheries & Wildlife, Bangor, ME

The 2005 range-wide status assessment of Eastern brook trout *Salvelinus fontinalis* by the Eastern Brook Trout Joint Venture (EBTJV) highlighted the need for extensive population status information from Maine. Subsequently, the Maine Department of Inland Fisheries & Wildlife (MDIFW) has undertaken a comprehensive, statewide effort toward updating pertinent information necessary for a contemporary species assessment. The MDIFW surveyed 1,990 stream sites in 435 6-level HUC sub-watersheds in 2007 and 2008. Survey sites were selected *a priori* according to probability of access in remote areas and spatially distributed within sub-watersheds to ensure representation of main stem, tributary, and headwater habitats. Standard protocols were used at all sites to assess fish community structure, stream habitat condition, and geomorphic stability. All fish encountered were accurately identified and counted; brook trout were weighed, measured, and sampled for scales and fin clips. Brook trout were found inhabiting 61.5% of all sites surveyed (1224/1990). Our objectives include producing a series of GIS data files for guiding conservation efforts and fishery management actions and implementing Maine's strategies for the EBTJV. Data products include statewide datasets of likely brook trout habitat, sub-watershed ranks for EBTJV priorities, and a preliminary stream condition index model for survey sites. Results to date are already directing conservation actions and habitat rehabilitation projects in Maine.

TROUT ANGLERS: A DEMOGRAPHIC DESCRIPTION AND ECONOMIC ANALYSIS

Anna Harris

Economist, Wildlife and Sport Fish Restoration Programs, U.S. Fish and Wildlife Service

In 2006, 6.8 million anglers fished for trout over 75 million days in the U.S. On average, anglers fish for trout 11 d per year. They spent US$4.3 billion and had a total economic output of $12 billion when indirect economic effects are factored in. The typical trout angler spends approximately $300 on trip-related expenditures and around $100 on trout fishing equipment. Participation varies widely from 871 thousand trout anglers in California to 14 thousand in Rhode Island. Trout anglers are predominately male (79 percent) and are well educated (13 percent earned graduate degrees). Resident trout anglers are willing to pay $56 per fishing day for their fishing experience while *out-of-staters* have a separate value of $135 per fishing day. Along with the willing-to-pay values, resource managers must also consider the potential impacts of climate change on trout populations and their habitats when evaluating management plans. This paper uses data from the 2006 National Survey of Fishing, Hunting and Wildlife-Associated Recreation focusing on participation and expenditures by U.S. residents 16 years of age or older, not including Great Lakes fishing. The results provide another tool for policy makers and resource managers to use when evaluating management actions that would have an impact on trout fishing.

Determining Detailed Spatial and Temporal Variability of Water Temperature Using Distributed Temperature Sensing to Evaluate Fish Habitat Suitability in a Great Basin Headwater Stream

Christine E. Hatch[a,b], David E. Prudic[a,c], Laura Belica[d], Scott W. Tyler[a,e]

[a] University of Nevada Reno, Geological Sciences & Engineering, MS175, Reno, NV 89557; (775) 784-1953

[b] chatch@unr.edu, (775) 784-4509

[c] prudic@unr.edu, (775) 784-5596

[d] Great Basin National Park, 100 Great Basin NP, Baker, Nevada 89311; laura_belica@nps.gov , (775) 234-7331 x232

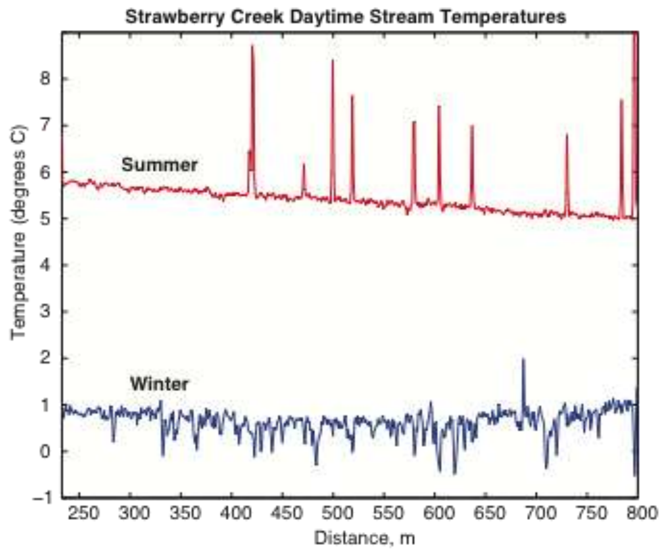
[e] styler@unr.edu, (775) 784-6250

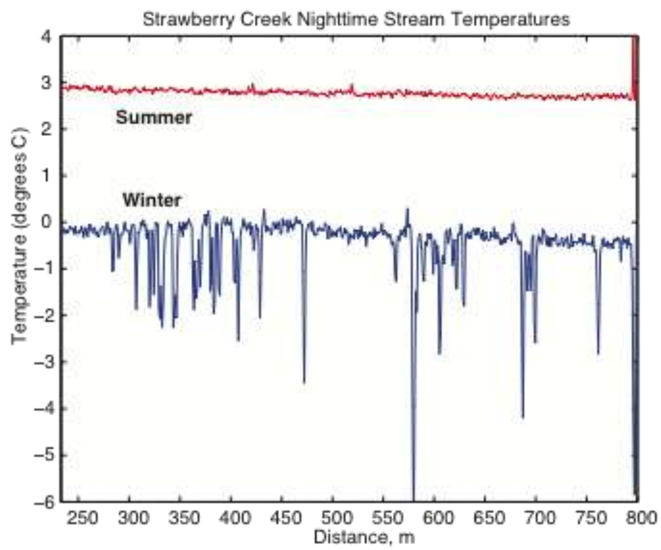
In semiarid regions such as the Great Basin, montane headwaters are often the only streams available for native trout restoration as lower elevation stream sections have been developed for water use. The temperature regime is a key ecological component of such streams. In addition to broad seasonal variations, headwater stream temperatures vary over small temporal and spatial scales in response to weather, diel cycles, groundwater discharge, channel morphology, riparian cover, and other factors. These temperature variations can affect habitat suitability at different times of year or life history stages and influence the distribution and relative abundance of fish populations.

Groundwater discharge is known to have a significant influence on stream temperatures, yet can be difficult to quantify due to its spatial and seasonal temperature variability. Distributed Temperature Sensing (DTS) offers a unique opportunity to obtain high-resolution stream temperatures over time and space, allowing for a detailed assessment of otherwise inaccessible habitats throughout the year. Distributed Temperature Sensing analyzes Raman-Spectra scattering of a laser-light pulse to infer distributed temperatures with ≤ 0.1°C accuracy, as often as every 10 s along each 1-2 m of a fiber-optic cable for up to 5 km in length.

A pilot project using the DTS method was initiated in December 2008 to provide information on variations in stream temperatures in a headwater considered for reintroduction of native fishes in Great Basin National Park. The method was tested along a 500-m long reach 2,440 m above sea level during three seasons: early summer runoff, late summer base flow, and late winter low-flow. These spatially and temporally continuous temperature data provide a better understanding of stream temperature variations along the channel through time, identify the spatial extent of the influence of upstream groundwater discharge on downstream temperatures, and aid in the identification of reaches likely to provide suitable winter habitats.

Strawberry Creek begins at a spring, and in the upper few kilometers gains flow from diffuse and discrete groundwater inflows throughout its narrow drainage. During runoff and late-summer base flow conditions, spring discharge comprises a significant fraction of overall streamflow in the upper reaches, maintaining stream temperatures at ≤ 6 °C even during the hottest hours of the day. In March of 2010, significant snowpack still covered the stream, and nighttime temperatures were well below 0 °C. During this period, the relatively warmer groundwater inflows and insulating capacity of the snow cover are crucial to providing suitable thermal conditions for trout survival over the winter. In Figure 1, a sample of stream temperatures from downstream ~230 m to upstream ~800 m are shown during (a) daytime and (b) nighttime, for the summer (red) and winter (blue) extremes. Large temperature excursions occur where the fiber optic cable was exposed to air and sun (summer) or snowpack (winter). During the summer, the stream warms downstream. In winter, the complex interaction between cold melt-water and warmer groundwater inputs create both warming and cooling trends along the reach, which identify thermally preferable stream habitat.


Strawberry Creek Daytime Stream Temperatures
Summer
Winter
Temperature (degrees C)
8
7
6
5
4
3
2
1
0
−1
250 300 350 400 450 500 550 600 650 700 750 800
Distance, m


Strawberry Creek Nighttime Stream Temperatures
Summer
Winter
Temperature (degrees C)
4
3
2
1
0
−1
−2
−3
−4
−5
−6
250 300 350 400 450 500 550 600 650 700 750 800
Distance, m

RESTORATION PROJECTS FOR NATIVE TROUT AND CHAR IN KOOTENAY NATIONAL PARK OF CANADA

S. Humphries

Aquatic Specialist, Yoho, Kootenay and Banff National Parks of Canada, Field, British Columbia

Kootenay National Park of Canada was established in 1920. It is located at the headwaters of the Kootenay-Columbia river system. This park is part of the contiguous Canadian Rocky Mountain Parks World Heritage site. The main stressors for the aquatic ecosystem are the current road network and Parks Canada's historic fish stocking practices (1939- 1976). In 2006, Parks Canada started two projects to advance aquatic restoration and remediation. Updated stream fish inventories were completed with genetic analysis for westslope cutthroat trout *Oncorhynchus clarkii lewisi* and a detailed culvert inventory was completed for all the road stream crossings in the park. The three low elevation lakes that historically contained cutthroat trout or bull trout *Salvelinus confluentus* are now primarily or exclusively dominated by introduced brook trout *Salvelinus fontinalis*. At least two high elevation, historically fishless lakes contain self sustaining introduced brook trout populations. The flowing waters still primarily contain native fish assemblages with a few hotspots of introduced brook trout. The genetics for the cutthroat trout are compromised throughout the park with only two locations showing 1% or less rainbow trout *Oncorhynchus mykiss* hybridization. Eighty culverts were located on streams where fish were either likely or confirmed. Of those 80 culverts, 55 are considered full barriers to at least some species or some life stages, an additional 13 are partial barriers. Only one full barrier culvert is isolating a pure cutthroat trout population, two other barriers may be strategically important for future restoration efforts. Many smaller scale stream or lake level restoration opportunities to protect or enhance bull trout and cutthroat trout exist. Three culvert remediations to restore fish passage are planned for 2010. One creek requires immediate brook trout suppression to preserve one of the last remaining pure cutthroat trout populations. Feasibility planning for two lake-stream fish restorations has begun.

DISTRIBUTION AND ABUNDANCE OF MIGRATORY ADULT BULL TROUT IN THE UPPER OLDMAN RIVER DRAINAGE, SOUTHWESTERN ALBERTA, CANADA

B.J. Hurkett

Alberta Conservation Association, 1609 3rd Ave. S, Lethbridge, Alberta, Canada, T1J 0L1, brad.hurkett@ab-conservation.com

Bull trout *Salvelinus confluentus* have been reduced to 34% of their historic distribution in the Oldman River watershed due to the effects of human activities. In particular, bull trout in the upper Oldman River (UOM) drainage are threatened by increasing pressure from industrial and recreational development. The Alberta Conservation Association is currently conducting a bull trout population assessment to determine the abundance and distribution of adult migratory bull trout within the UOM drainage. Our study focuses on intercepting post-spawn migratory bull trout using conduit fish traps stationed in key spawning tributaries. We marked adult bull trout ≥300 mm fork length with internal transponder tags and used recapture data to monitor spawning frequency and track migration movements of individual fish. Following 2010 trapping efforts, recapture data will also enable us to determine fish abundance using the Jolly-Seber open-population capture-recapture model. To compliment trap data, we conducted redd counts in all suspected spawning tributaries in the UOM drainage to determine redd densities and identify new spawning habitats. Since the project began in 2007, 280 adult bull trout have been tagged in the UOM drainage. We intercepted 200 (72%) of these fish in Hidden Creek, a key drainage for spawning bull trout in the UOM drainage which has been slated for commercial logging. Recapture data from elsewhere in the drainage links another 15 adults to Hidden Creek; reinforcing its importance as a critical spawning tributary for migratory bull trout throughout the drainage. Our study, which concludes in 2010, has resulted in an indefinite deferral of logging operations in the Hidden Creek drainage until protective reclassification of the spawning tributary is complete. To date, Hidden Creek angling regulations have also been amended to prohibit angling prior to the spawning season. These results highlight the importance of basic life history and habitat-use information for effective management of fishery resources.

WESTSLOPE CUTTHROAT TROUT RESPONSES TO LANDSCAPE DISTURBANCE IN THE UPPER OLDMAN RIVER WATERSHED, ALBERTA, CANADA

John Jackson

AMEC Earth & Environmental, Aquatic Science Group, Calgary, Alberta, Canada.

Watershed-scale assessments are useful tools for describing the effects of land use on the abundance and distribution of North American salmonid species. We conducted a watershed-scale analysis in four headwater catchments of Alberta's Oldman River to evaluate the effects of land use and natural drainage attributes on westslope cutthroat trout *Oncorhynchus clarkii lewisi*. A blocked regression, using catchments as fixed factors, was applied to test relationship significance between response and test variables and account for the discrepancy in fish productivity between catchments. Significant negative relationships between westslope cutthroat trout abundance and land-use variables were identified (e.g., trout biomass and stream-crossing density). However, the strongest regressions were positive relationships between trout abundance and natural drainage features - drainage area (a measure of surface drainage) and confluence distance (a measure of position in the watercourse). We conclude that incorporating these natural drainage attributes into watershed-scale assessment modeling is warranted and is particularly important in the assessment and management of eastern slopes headwater catchments.

EVALUATION OF A SAMPLING APPROACH TO MONITOR THE STATUS OF GREAT BASIN REDBAND TROUT IN EASTERN OREGON

Steven E. Jacobs, Stephanie A. Miller, Stephanie L. Gunckel, and Shannon Richardson

Oregon Department of Fish and Wildlife, Corvallis Research Lab, Native Fish Investigation Project, 28655 Hwy 34, Corvallis, OR

Redband trout *Oncorhynchus mykiss newberrii* inhabit arid environments ranging from montane forests to desert shrub and grasslands in streams where extreme fluctuation in flow and temperature are common. Hydrological cycles of flood and drought, paired with increased anthropogenic disturbance of already naturally fragmented habitat have prompted concern over the status of desert trout populations. The summer 2009 field season marked the completion of the third of a 6-year sampling effort to assess the distribution and abundance of redband trout in the six interior basins of Oregon's high desert: Catlow, Chewaucan, Fort Rock, Goose Lake, Malheur, and Warner Valley Species Management Units (SMUs). Across all sampling years, sites were randomly selected using Generalized Random Tessellation Stratified (GRTS) design developed by the EPA which provides a random spatially balanced sample allowing for statistically rigorous evaluation of status, trend, and distribution at multiple spatial scales. A total of 699 site surveys were conducted over the course of the study covering nearly 2% of the entire 2,420-km sampling frame. Populations of age+1 redband trout at the SMU level have remained viable and relatively stable since they were first intensively sampled in 1999 and throughout the course of this study. Estimates of overall landscape-wide average abundance of age+1 redband trout was of similar magnitude and had comparable precision across all three study years, averaging 1,116,937 ± 18%. However, abundance at the SMU and population level showed substantial variation, both spatially and interannually. Site level fish densities (fish/m) sampled at repeatedly visited annual sites (2007-2009) show significant differences between years, specifically between 2007 and 2009 in the Chewaucan and Malheur SMU's. Target levels of relative precision were not achieved at the SMU level, and infrequently at the population level. Increasing the number of sites sampled to increase precision is not likely, given limited funding. Yet, the current study design falls short of providing precise information at the population level reducing the data available to develop a conservation management plans. Alternative sampling designs that would maximize data acquisition at the population level while allowing for estimates of yearly variation were explored and suggested.

ESTIMATING AN EFFECTIVE POPULATION SIZE BY PEDIGREE RECONSTRUCTION FOR BROOK TROUT POPULATIONS IN CONNECTICUT HEADWATER STREAMS

Yoichiro Kanno[1], Jason C. Vokoun[1], and Ben H. Letcher[2]

[1]Department of Natural Resources and the Environment, University of Connecticut
[2]U.S. Geological Survey, Conte Anadromous Fish Research Center

Brook trout *Salvelinus fontinalis* have been declining in much of its native habitat range in eastern North America, and its populations are typically relegated to small headwater streams in southern New England. We used pedigree reconstruction to infer an effective population size in two select headwater network streams in Connecticut. Brook trout were captured via backpack electrofishing from continuous stream stretches in the two study streams (channel network lengths of 7.5 km and 4.5 km) during the summer of 2008, and eight neutral microsatellite loci were genotyped for over 1,000 individuals from the two systems. Genetic analysis indicated that both males and females were polygamous, although single pair matings were also inferred. This complex mating system resulted in many half-sib dyads, and it may have helped maintain high genetic diversity. Very few large-sized full-sib families (>3 individuals) were inferred in the samples, indicating that environmental pressure had acted similarly across many families. When large-sized full-sib families were identified among juveniles (80-140 mm), individuals of the same full-sib families were often distributed in a spatially clustered manner. Using a sibship assignment method (Wang 2009), the effective population size was estimated at 210 (95% CI: 172-259) and 91 (95% CI: 67-123) in the two study sites, which translates into about 2-3 successful breeders per 100 m. This study successfully inferred mating strategies of brook trout based on genetic data and provided an insight into a mechanism of population persistence in headwater streams.

ECOLOGICAL AND GENETIC INTERACTIONS BETWEEN HATCHERY AND WILD STEELHEAD IN EAGLE CREEK, OREGON

Maureen Kavanagh[1], Bill Brignon[1], Doug Olson[1], Susan Gutenberger[2], Andrew Matala[3], William Ardren[3]

[1]U.S. Fish and Wildlife Service, Columbia River Fisheries Program Office, 1211 S.E. Cardinal Court Suite 100, Vancouver, WA 98683, maureen_kavanagh@fws.gov
[2]U.S. Fish and Wildlife Service, Lower Columbia River Fish Health Center, Willard, WA.
[3]U.S. Fish and Wildlife Service, Abernathy Fish Technology Center, Longview, WA

Eagle Creek National Fish Hatchery spawns and raises juvenile coho salmon *Oncorhynchus kisutch* and juvenile steelhead trout *Oncorhynchus mykiss* that are released into Eagle Creek within the Clackamas River Basin. The hatchery operates within the confines of the Endangered Species Act (ESA); however, limited information exists on the ecology and biology of wild ESA listed fish in Eagle Creek. From 2005 to 2009, we evaluated the ecological and genetic interactions between a hatchery stock of steelhead raised at Eagle Creek National Fish Hatchery for harvest and a U.S. Endangered Species Act threatened steelhead population in Eagle Creek, Oregon. We monitored adult returns, hatchery smolt releases, abundance and habitat selection of juvenile fish, and genetic structure of hatchery and wild origin steelhead. From our findings, we concluded that hatchery steelhead may pose an ecological risk to wild steelhead in Eagle Creek, particularly at the juvenile-smolt stage. We found that hatchery residuals made up approximately 9% of the yearling stream population and 1% of the sub-yearling stream population of steelhead. Although no displacement of wild fish was documented, the abundance of hatchery residuals may still pose a risk to the wild steelhead in Eagle Creek. Additionally, genetic analyses indicated that in some years natural production in upper Eagle Creek was influenced by natural spawning of hatchery steelhead. When evaluating the program at Eagle Creek, we tried to consider the full biological, social, and economic value the steelhead fishery brings to the Clackamas River Basin. The recommendations from this study are designed to improve hatchery and genetic management practices, assist with conservation and recovery of ESA listed populations, and contribute to recreational, commercial, and tribal harvest.

THE ECONOMIC IMPACTS AND BENEFITS OF RESTORATION-BASED IMPROVEMENTS TO MONTANA'S BLACKFOOT RIVER SPORT FISHERY

Joe Kerkvliet

Professor Emeritus, Oregon State University and Senior Resource Economist, The Wilderness Society, Bozeman, Montana, 59715, phone 406 581 9826

We estimate the economic impacts and economic benefits of an improved sport fishery resulting from ecological restoration actions affecting Montana's Blackfoot River. We conducted angler surveys on the Blackfoot main stem, North Fork, and Lander's Fork during 2009. We obtained data on over 200 anglers' expenditures, fishing trips, and fishing days. We also obtained contingent behavior data on trip and day changes if restoration work led to improved fishing in terms of increased numbers and/or size of trout and chances of catching (and releasing) a trophy bull trout *Salvelinus confluentus*. Empirical analysis suggests that Blackfoot anglers would spend an additional US$7 million per year if average trout size were to increase 20%. Estimated economic value increases by $4 million if the chance of catching a trophy bull trout were to double.

INHERITANCE OF LAKE TROUT MICROSATELLITE LOCI: IMPLICATIONS FOR CONSERVATION GENETICS AND HATCHERY PRODUCTION

Benjamin Kissinger

Fisheries and Biology, University of Wisconsin – Stevens Point, 1908 College Ave, Stevens Point, WI 54481

Restoration of Great Lakes lake trout *Salvelinus namaycush* currently relies on hatchery propagation to establish and maintain populations. As with any program, monitoring and evaluating the efficacy of management efforts is a focus of the propagation program. The use of genetic markers can provide cost-effective, accurate evaluation of strain-specific success over multiple generations. To achieve such goals, an effective suite of molecular markers is required. Ideal genetic markers should be codominant (i.e., observed alleles from both mother and father) and inherited in a Mendelian pattern (i.e., strict inheritance of alleles from parents). My objectives are (1) determine if a suite of 11 lake trout-specific microsatellite genetic markers are inherited in a codominant, Mendelian fashion; and (2) determine if any of the 11 genetic markers are significantly linked to observed albinism in lake trout. To test the codominant Mendelian nature of these markers, lake trout broodfish and a sample of resulting offspring (N=90/cross) from the Ontario Ministry of Natural Resources' Codrington Fish Hatchery were genotyped for 11 microsatellite loci. Given the known parental genotypes, the error rate in inheritance and mutation rate of the loci was estimated. This was replicated for six total parental crosses and the observed allele frequency distribution was compared to expected distribution under a Hardy Weinberg model using chi-square tests. Two of the six tested crosses produced a 50:50 ratio of normal to albino fish. No parental fish exhibited albino phenotypes. This is consistent with a single-gene albinism control (albino allele is recessive) with each of the two parents being heterozygotes for the albino trait. Allele identities and frequencies were compared for correlation to the albino phenotype. The results of this study have strong implications for Great Lakes lake trout propagation.

FISHERIES RESTORATION POTENTIAL OF THE CLARK FORK RIVER SUPERFUND SITE: HABITAT USE AND MOVEMENT IN RELATION TO ENVIRONMENTAL FACTORS

Mariah P. Mayfield[1*], Thomas E. McMahon[1], Pat Saffel[2], Trevor M. Selch[3], Jason Lindstrom[1], and Brad Liermann[1]

[1]Fish and Wildlife Program, Department of Ecology, Montana State University, Bozeman, MT;
[2]Montana Fish, Wildlife, and Parks, Region 2, Missoula, MT;
[3]Montana Fish, Wildlife, and Parks, Helen, MT
[*]Corresponding author: mariah.mayfield@gmail.com

Fish populations in large portions of the Upper Clark Fork River were decimated in the 1900s due to the deposition of hazardous mining waste. Improvements in wastewater treatment and mine waste remediation have resulted in a rebound of trout populations, although in numbers well below expected carrying capacity. Nonnative trout (primarily brown trout *Salmo trutta*) now dominate the fish assemblage, likely due to their greater ability to tolerate heavy metal contamination still present in the system. In 2009 and 2010, 200 trout were surgically implanted with radio transmitters throughout the Upper Clark Fork River, from Warm Springs to the confluence with the Blackfoot River. Transmitters were spatially distributed evenly throughout the study area and species were selected based on relative abundance found in the river. The primary species tagged was brown trout *Salmo trutta*, although westslope cutthroat trout *Oncorhynchus clarkii lewisi*, rainbow trout *O. mykiss*, suspected cutthroat trout-rainbow trout hybrids, and bull trout *Salvelinus confluentus* were also tagged in reaches where they were present. Radio-tagged fish are relocated at least once a week during spring, summer, and fall (more during periods of spawning) and at least twice per month during the winter, until winter 2011. Using relocation data, we will identify areas of critical habitat, such as spawning areas, refuge habitat, and overwintering habitat. Movement patterns will also be analyzed to assess how fish are reacting to the environmental factors unique to the Upper Clark Fork River Basin (combination of heavy metal pulses during storm events and warm summer temperatures). We will also determine whether native trout species are more sensitive to poor water quality conditions than nonnative trout. By identifying the critical habitat of the system, efforts can be made to guide remediation to be most effective for overall fisheries restoration and preservation of native trout in the system.

DEVELOPING AND SUSTAINING INTEREST IN WILD TROUT

Pete Segerson

Field Operations Supervisor, West Central RFO, Wisconsin Department of Natural Resources, phone: (715) 284-1447, fax: (715) 284-1737, e-mail: Peter.Segerson@wisconsin.gov

The Wisconsin Department of Natural Resources West Central Region has a Cooperative trout rearing program with 60+ years of history. The original focus of the program was rearing domestic strains of brown trout Salmo trutta and rainbow trout Oncorhynchus mykiss for stocking catchable size trout. More than 20 individual rod and gun clubs developed ponds, raceways, and spring heads to rear trout.

In the 1990s Wisconsin DNR began experimenting with propagating feral strains of trout for re-introduction into restored waters. In 1990 the co-op trout program produced 56,000 nonnative domestic strain trout. By 2000 (20%) of the club-raised trout were feral strain. In 2010 (84%) of the trout being raised are feral strain and 60% of those are native brook trout Salvelinus fontinalis.

Land use changes, a greatly improved habitat enhancement and restoration program, and stocking feral strain trout has added 260 mi of newly classified trout stream to Wisconsin waters in the last decade.

The clubs did not immediately buy into the idea of replacing domestic strains with wild trout. The 21 clubs have approximately 1,800 members. Many of the clubs involve school groups, and it is estimated about 400 students hear the message of wild trout and their habitat from the co-op trout program annually. Most of the clubs are now also funding and helping with habitat restoration, and some have taken public fishing easements.

One case history will also be presented where Dutch Creek suffered a manure-spill-related fish kill estimated at 90% mortality of the brown trout. Feral brook trout being raised by a co-op pond were used to repopulate the stream. Results of that stocking will be demonstrated.

The upshot of the co-op trout stocking program has been to interest angler groups and students in the value of wild trout and the places where wild trout live.

GENETIC TYPING OF WILD BROOK TROUT IN NORTH CAROLINA

Jacob M. Rash[1], Douglas A. Besler[1], and Amanda M. Bushon[2]

[1]North Carolina Wildlife Resources Commission. 645 Fish Hatchery Road, Marion, North Carolina 28752

[2]North Carolina Wildlife Resources Commission, 20830 Smokey Mountain Expressway, Waynesville, North Carolina 28786

The brook trout *Salvelinus fontinalis* is the only salmonid native to North Carolina. Anthropogenic alterations to the landscape and introductions of nonnative salmonids have greatly reduced its range. Intensive stockings of northern strain brook trout also diminished the genetic integrity of many native Southern Appalachian strain brook trout populations. The North Carolina Wildlife Resources Commission (NCWRC) has been involved in a long-term effort to identify and genetically type wild book trout populations within the state. To date, 589 wild brook trout populations have been identified and of these, 478 have been genetically typed via allozyme analysis. Results from testing indicate that 38% of the populations are Southern Appalachian origin, 10% are northern origin and 52% are of mixed genetic origin. Microsatellite DNA analysis will be employed by the NCWRC to gain further insight into the historic distribution of Southern Appalachian brook trout, examine current population relatedness, and develop a genetically-based restoration framework. Continuing protection of existing Southern Appalachian brook trout populations, and the restoration of those extirpated, can only be achieved if managers have a firm understanding of the genetic variance associated with the species.

USE OF MICROSATELLITE DNA ANALYSIS TO CHARACTERIZE BULL TROUT SPAWNING STOCKS IN THE NORTH SASKATCHEWAN RIVER DRAINAGE, ALBERTA, CANADA

Mike Rodtka[1] and Chad Judd[2]

[1]Alberta Conservation Association, Rocky Mountain House, Alberta, Canada, Mike.Rodtka@ab-conservation.com, Phone: 403-845-8234

[2]Alberta Conservation Association, Rocky Mountain House, Alberta, Canada, Chad.Judd@ab-conservation.com, Phone: 403-845-8371

Following an approach developed by the United States Fish and Wildlife Service, Alberta recently adopted use of core areas to assess bull trout *Salvelinus confluentus* populations. Identification of the Middle North Saskatchewan River core area, which includes the Ram River and Fall Creek, relied heavily upon expert opinion and anecdotal evidence. No formal evaluation of the bull trout stock(s) within the area had occurred until our study in 2007 — 2009. Based on angler reports, Fall Creek was suspected of being an important spawning stream for migratory bull trout within the core area. In addition to telemetry and migration trapping, we used microsatellite DNA analysis to evaluate the relatedness of the Fall Creek stock to other putative migratory stocks in the area. We collected tissues samples (adipose fin clip) from bull trout ≥ 200 mm fork length throughout the core area for comparison to samples collected from Fall Creek. Tissue samples were analyzed by Eric Taylor at the University of British Columbia. Only collections from Fall Creek ($n = 50$) and the Ram River ($n = 41$) resulted in sufficient sample sizes for rigorous testing. Nine microsatellite loci were assayed to test for population differentiation and run population assignment tests. The Ram River and Fall Creek stocks were significantly different from one another genetically ($\Theta = 0.038$, $P < 0.001$) and only a single fish was rejected as a member of the population from which it was sampled (Fall Creek; $P < 0.001$), indicating a high degree of spawning site fidelity. These results, which indicate that at least two spawning populations of bull trout occur within the core area, underscore the utility of microsatellite DNA analysis for studies such as ours, particularly when combined with more traditional approaches such as telemetry and migration trapping.

PRAIRIE RIVER SPECIAL TROUT REGULATIONS — DID THEY WORK?

David A. Seibel[1] and Jean V. Adams[2]

[1]Wisconsin Department of Natural Resources
[2]United States Geological Survey

Studying the effects of special trout regulations helps fisheries managers tailor regulations that may better mesh resource potential with anglers' desires and expectations. The Prairie River fishery is fully supported through naturally reproducing brook trout *Salvelinus fontinalis* and brown trout *Salmo trutta* populations. Special regulations (high size limits, reduced daily bag limits, and artificial-lure-only restrictions) were in effect on a section of the Prairie River for 5 years from 2003 to 2007. They were intended to improve numbers of larger trout and overall size quality.

Double-run, population estimating electrofishing surveys were done for eighteen consecutive years (1992-2009) on a survey station within this section of the Prairie River. Eleven of the survey years were prior to the special regulations being in place, five were during the special regulation years, and 2 were after the special regulations had been removed. Data from these surveys were analyzed to determine if the special regulations worked as intended and increased the numbers of larger trout. Analyses found that numbers of 10-inch and larger and 11-inch and larger brook trout were significantly higher with special regulations in place (Two-Sample T- Tests; $P < 0.05$). Numbers of larger brown trout did not increase with special regulations. In fact, several size groupings of brown trout (8-inch plus, 10-inch plus, 12-inch plus) significantly decreased under special regulations (Two-Sample T-Tests; $P < 0.05$).

Analysis of different time periods suggests that it took about 1 to 2 years for brook trout to start to "grow into" the higher size limit before positive results initially became evident. The data also suggest that 5 years of special regulations is not long enough for the full effects of the regulations to be realized. This may be part of the reason why longer lived brown trout did not exhibit a positive response to the special regulations in this study. These analyses also suggest that when the special regulations were removed, the improvements to the brook trout size structure were not immediately "fished down". Almost two full fishing seasons after the regulations were removed, the numbers of 10-inch and larger and 11-inch and larger brook trout were not significantly different from the last 3 years under special regulations (the 3 best years). The numbers did come down, but the means were closer to the means under the special regulations and 2 to 4 times higher than the means prior to the special regulations.

Analyses of a second survey station within the special regulations zone with intensive habitat improvements (channel shaping, wing deflectors, boom covers, boulders, half-logs) suggest that habitat improvements were more effective at improving size structure of both brook trout and brown trout than the special regulations.

TITLE: PREDICTING THE PROGRESS OF AN INVASION – ACTIVE DISPERSAL OF NEW ZEALAND MUD SNAILS UNDER VARIABLE HYDROLOGICAL AND RESOURCE CONDITIONS

Laurie Marczak and Adam Sepulveda

Affiliation: University of Montana, College of Forestry, 32 Campus Drive, Missoula, MT 59812-0004, (406) 243-2932, adam.sepulveda@mso.umt.edu

New Zealand mud snails (NZMS; *Potamopyrgus antipodarum*) have spread rapidly across the western United States, but little is known about mechanisms that drive their spread within an invaded stream. We used a field experiment to test if upstream movement is a potential vector of NZMS spread and if movement is modified by flow velocity and resource availability. We found that flow velocity influenced movement direction and rate, while resource availability influenced the number of individuals that moved. In slow-flow treatments, individuals moved upstream at faster rates than previously recorded. Upstream movement rates approached 3 m/h. In fast-flow treatments, most individuals were dislodged downstream and upstream movement rates were less than 2 m/h. In low-resource treatments, individuals were more likely to move away from their initial starting locations. We suggest that upstream movement may be important in establishing new populations and that increases in flow velocity may be an effective means to slow the upstream spread of NZMS. The surprisingly fast movements that we recorded suggest that factors other than NZMS movement rate limit population spread.

Implications of Stocking with Brood Fish to Management with Resident Brown Trout Stock in the Gradac River

Predrag Simonović, Slavica Grujić, and Vera Nikolić

University of Belgrade, Faculty of Biology, Studentski trg 16, 11000 Belgrade, Serbia

The Gradac River (a tributary of the Sava River in the Danube River drainage area of Serbia) supports a reproducing population of brown trout *Salmo trutta* and is managed under catch-and-release fly-fishing-only regulations. Brown trout parr are stocked on occasion when strong spring torrents wash out newly hatched brown trout fry. In spring 2008 and 2009, there were unauthorized stockings of about 100 brood-size (about 1 kg) brown trout in the upper section of the Gradac River. In summer 2009 we sampled the brown trout population in the upper and lower (unstocked) section of the river and compared population statistics to data collected in summer 2003. In the upper stocked section, the age and size at maturation of brown trout increased as revealed by breakpoint values obtained from Piecewise Linear Regression. Brown trout density decreased significantly as did biomass and production after stocking. No changes in population statistics were found in the lower unstocked section. We concluded that stocking these large brood fish had an adverse effect on the resident brown trout.

Creation of a Captive Wild Strain of Brook Trout in Southwestern Wisconsin

Richard S. Stewart and Robert Fahey

Wisconsin Department of Natural Resources, 3911 Fish hatchery Road, Fitchburg, WI53711

Fisheries managers in Wisconsin have been interested in managing trout streams with trout propagated from wild parents for nearly 20 years. Wisconsin began experimenting with the stocking of trout derived from wild parents in 1994. Studies in the state concluded that fish raised from wild parents have better survival, and that if reproduction begins to occur in streams that the resulting genetics of the populations are more diverse. Initial efforts to use wild fish in Wisconsin propagation collected fish from the wild and held them until the fish were ripe. Hatchery managers spawned these fish weekly until the spawning period was complete. Propagation efforts following this procedure were followed for both brown trout *Salmo trutta* and brook trout *Salvelinus fontinalis*. However, as successful results of the wild stocking were realized, more managers desired to use the wild progeny. Demand for wild fish for stocking became greater than a reasonable effort of collecting parents to produce eggs could provide. In addition, recent increased concern over introduction of disease into hatcheries became an additional factor of concern with bringing wild parents into a hatchery. These factors in combination have led us to develop captive brood stocks directly from wild parents. We summarize production of eggs by wild parents held until they are ripe and then spawned. This technique was insufficient in producing enough progeny to meet fisheries manager's demand. We also summarize the creation of a captive "wild" brood stock kept in southern Wisconsin. This second model has allowed us to produce all of the eggs and progeny required to meet managers' needs in southwestern Wisconsin.

Libraries as Partners in Research and Maintaining and Developing a Trout and Salmonid Collection

James Thull

Assistant Professor/Reference Librarian, Montana State University- Bozeman, jjthull@montana.edu, 406-994-5305

I am proposing a poster session for the Wild Trout Symposium on the value of libraries and librarians as a resource and information on building and maintaining a trout and salmonid library. I will highlight the Montana State University Libraries Trout and Salmonid collection and its purpose and value to researchers, both local and from afar. I will explain what librarians can offer in terms of research assistance and dissemination of knowledge and touch on other related collections in existence and how MSU is working with them to make our common pool of knowledge larger and our individual collections stronger.

Riverbed Manipulation Techniques to Restore and Enhance Wild Trout Fisheries

Joseph Urbani

Fisheries Biologist/Principal, Company: Urbani Fisheries, LLC, 198 North Star Lane, Bozeman, MT 59718, (406) 587-0588, jurbani@urbanifisheries.com

This presentation will discuss methods used to restore and create wild trout habitat in two select rivers. Also discussed are techniques for repairing and stabilizing banks that are experiencing active surface erosion or channel cutting processes. The main objective is to design and enhance every aspect of habitat that trout require, as well as improve the entire aquatic resource to achieve the highest biological and recreational values. This approach to optimizing trout habitat in rivers is based on a set of techniques referred to as “bed manipulation.” This involves reorganizing the native elements in the stream to favor trout and their food base organisms, thereby allowing the stream to reach its full potential as a trout fishery. Working with the natural characteristics of the stream, it optimizes habitat conditions for trout. Typically, pools are located at a point in the channel where the gradient and velocity are within parameters such that the natural scouring force of the flowing water will maintain depth, while the overall pool habitat will remain stable through time and various flow regimes. Materials are excavated from the channel to form a pool, placing the material on the opposite bank creating a point bar. A point bar has the effect of narrowing the channel and increasing velocity which increases the ability of the stream to transport sediments through the pool, as well as creating a helical flow pattern that scours the pool thereby maintaining depth. Excavated gravels are also placed at another point in the channel (tail outs) where conditions are conducive to trout spawning. These types of in-stream enhancements, combined with bank stabilization and riparian restoration techniques, have been implemented on multiple sections of the Pecos River in New Mexico and the Musconetcong River in New Jersey, whose programs were funded both privately and federally.

Effectiveness of Isolation Management as a Conservation Strategy for McCloud River Redband Trout in Two Northern California Streams

Margaret A. Wilzbach and Roman G. Pittman

USGS California Cooperative Fish Research Unit, Humboldt State University, Arcata CA 95521, wilzbach@humboldt.edu; 707-826-5645

McCloud River redband trout *Oncorhynchus mykiss* ssp. are believed to be a relict subspecies of non-anadromous rainbow trout that evolved in isolation in the upper Sacramento River drainage in northern California. Genetic integrity of the fish is threatened by introgressive hybridization with hatchery rainbow trout, and populations are at risk from invasions of nonnative salmonids and habitat loss. High conservation value deriving from vibrant body coloration, its adaptations to harsh, fragmented environments, and its history of isolation have made the trout the focus of a multiple signatory conservation agreement to prevent its listing under the Endangered Species Act. To evaluate the potential effectiveness of isolation management to protect and restore populations in Trout and Tate creeks within the established McCloud redband trout refugium, we located existing and potential barriers to fish movement, evaluated resource availability, and estimated minimum stream lengths required to maintain genetically viable population sizes of 2,500 individuals from reach-scale abundance and survival estimates in two refuge streams. Trout Creek was judged to be a poor candidate for deliberate isolation because a percolation barrier isolates it from the upper McCloud River and further barriers would fragment already limited habitat. Although trout from the lower reach of Tate Creek showed evidence of hybridization, the stream supported a higher trout density and available habitat exceeded estimated minimum stream length (7.3 km); as such it may represent a viable isolation candidate, with sufficient resources to support growth of translocated populations. Easing of state permitting restrictions for scientific research would allow similar evaluation of risks and benefits of isolation management to protect the trout in other streams within the McCloud River refugium.

LIST OF PARTICIPANTS

This list is generated from the participants registered prior to the start of the symposium. It does not include participants who joined the symposium as a walk-in registration.

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
James	Baxter	103-333 Victoria Street		Nelson		V1L 4K3	250-352-6874	James.Baxter@bchydro.com	Senior Fisheries Biologist	BC Hydro/Fish and Wildlife Compensation Program
Jason	Blackburn	231 24 St South.	Lethbridge, Alberta, Canada	Lethbridge		T1J 3N2	403 381 5335	jason.blackburn@ab-conservation.com	Fisheries Biologist	Alberta Conservation Association
Rob	Buffler	1240 Railway Ave	Unit 200	Canmore AB, Canada		T1W 1P4	403 609 2666	marlis@y2y.net	Executive Director	Yellowstone to Yukon Conservation Initiative
Jenny	Earle	33 Citadel Bluff Close NW		Calgary, Alberta		T3G 5E1	403-851-2211	Jennifer.Earle@gov.ab.ca	Ms.	Alberta Fish and Wildlife
Kaare Manniche	Ebert	Danmarks Sportsfiskerforbund	Skyttevej 4	Bredsten, Denmark		7182	004575820699	kme@sportsfiskerforbundet.dk	Cand. Scient. biol.	Danish Sportsfisher Association
Paul	Harper	944 - 13th Street South		Lethbridge, Alberta		T1J 2X2	403-394-2927	paul.harper@dfo-mpo.gc.ca	Fish Habitat Biologist	Fisheries and Oceans Canada
Shelley	Humphries	Box 99		Field, British Columbia		VOA 1G0	250.343.6108	shelley.humphries@pc.gc.ca	Aquatic Specialist	Parks Canada
John	Jackson	6004	33rd Avenue NW	Calgary		T3B 1K3	403-387-1671	john.jackson@amec.com	Aquatic Biologist	AMEC Earth & Environmental
Finn Sivebaek	Jensen	Vejlsoevej 39		Silkeborg		DK-8600	+4521792195	fs@aqua.dtu.dk	Biologist	DTU Aqua - National Institute of Aquatic Resources
Lene	Jensen Scheel-Bech	NYROPSGADE 30		COPENHAGEN, DENMARK		DK-1780	+4572185870	LEJ@FD.DK	HEAD OF SECTION	DANISH DIRECTORATE OF FISHERIES
Chad	Judd	PO Box 336,	4919-51 Street, 1st Floor Provincial Building	Rocky Mountain House,Alberta, Canada		T4T 1A3	403-845-8371	chad.judd@ab-conservation.com		Alberta Conservation Association
William	Linklater	PO BOX 1170		Magrath, Alberta		T0K1J0	4035938396	hiddenwolf213@hotmail.com	Student	University of Lethbridge
Reginald	Madden	Reginald Madden	91 Beeches Road	Pictou, Nova Scotia, CAN		B0K1H0	902-396-7879	maddenrj@gov.ns.ca	Biologist	Nova Scotia Department of Fisheries & Aquaculture
Frank	Nielsen	Danmarks Sportsfiskerforbund	Skyttevej 4	Bredsten		7182	+45-40338284	fn@sportsfiskerforbundet.dk	Manager, MSc	Danish Sportsfisher Association
Jan	Nielsen	Vejlsoevej 39		Silkeborg		DK-8600	004521685643	janie@aqua.dtu.dk	Biologist	DTU Aqua - National

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
										Institute of Aquatic Resources
J Gary	Norris	310 Uganda Avenue	Victoria, B.C., Canada V9A 5X7	(Victoria)		(V9A 5X7)	250-381-6076	jgnorris@Shaw.ca	None	None (private citizen)
Mike	Rodtka	Box 336	Provincial Building, 4919-51 street	Rocky Mountain House, Alberta		T4T1A3	403-845-8234	mike.rodtka@ab-conservation.com	Biologist	Alberta Conservation Association
Predrag	Simonovic	University of Belgrade, Faculty of Biology	Studentski trg 16	Belgrade		11000	+381 63 864 9340	pedja@bio.bg.ac.rs	Associate Professor	Univ. Belgrade, Faculty of Biology
Eric	Valdal	441 Columbia St.	Kamloops, BC, Canada V2C 2T3	Kamloops, BC, Canada V2C 2T3		########	250-828-4354	eric.valdal@gov.bc.ca	Land Information Specialist	BC, Integrated Land Management Bureau
Simone	Vincenzi	Viale GP Usberti 33/A		Parma, Italy		43100	8313345592	simone.vincenzi@nemo.unipr.it	Mr	University of Parma
Nate	Catterson	712 Ocean Cape Rd		Yakutat	AK	99689	907 784 3359	ncatterson@fs.fed.us	Fisheries Biologist	USFS Yakutat Ranger District
Sarah	O'Neal	6601 Chevigny St		Anchorage	AK	99502	360-918-4352	sarahlouiseoneal@ak.net	Associate Director	Fisheries Research and Consulting
Paul	Skvorc	P.O. Box 871010		Wasilla	AK	99687	(907)301-9995	pskvorc@biopar.com	Mr.	Biopar
Samuel	Snyder, PhD	819 Delaney Street		Anchorage	AK	99501	907-903-5811	snyderaway@gmail.com	Assistant Director	Renewable Resources Coalition
Daniel	Magoulick	Department of Biological Sciences	1 University of Arkansas	Fayetteville	AR	72701	479-575-5449	danmag@uark.edu	Professor and AUL	USGS Coop Unit, Univ of Arkansas
Roger	Bloom	1701 Nimbus Road Suite C		Rancho Cordova	CA	95670	916 464-6355	rbloom@dfg.ca.gov	Heritage & Wild Trout Program Leader	California Department of Fish and Game
Walter	Duffy	Coop Fish & Wildlife Research Unit	Humboldt State University	Arcata	CA	95521	707-826-5644	walter.duffy@humboldt.edu	Dr.	USGS
Charles	Hammerstad	780 Portswood Dr		San Jose	CA	95120	408-997-5460	chamerstad@aol.com	Conservation Chairman	Flycasters of San Jose
David	Lentz	830 S Street		Sacramento	CA	95811	916 445-3773	dlentz@dfg.ca.gov	Native Trout Conservation coordinator	California Dept. of Fish and Game

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
Kathleen	Matthews	Pacific Southwest Research Station	800 Buchanan St	Berkeley	CA	94710	5105596454	kmat-thews@fs.fed.us	Research Scientist	USDA Forest Service
Stephanie	Mehalick	401 15th Street	Apt A	Sacramento	CA	95814	916-464-6358	smeha-lick@dfg.ca.gov	Fisheries Biologist	Pacific States Marine Fisheries Commission
Dan	Plummer	31725 Sea Level Dr		Malibu	CA	90265	310-699-4095	catskill-dan@mac.com	Vice President	Code Blue
Marty	Seldon	1146 Pulora Court		Sunnyvale	CA	94087-2331	408-736-5631	mmsel-don@sbcglobal.net	FFF Senior Advisor	Federation of Fly Fishers, WT-X Org Committee
Jeff	Weaver	2585 Bryan Way		Placerville	CA	95667	916-708-2425	jweaver@dfg.ca.gov	Associate Fisheries Biologist	CA Dept. of Fish and Game
Margaret	Wilzbach	Humboldt State University	1 Harpst St.	Arcata	CA	95521	707-826-5645	wilz-bach@humboldt.edu	Dr.	USGS California Cooper-ative Fish Research Unit
Douglas	Fruge	PO Box 25486-DFC		Denver	CO	80225	303-236-4215	doug_fruge@fws.gov	Program Supervisor	Fisheries Program, US Fish & Wildlife Service
Anders	Halverson	802 Pine St		Boulder	CO	80302	3035207009	andershalver-son@xsmail.com	Research Associate	University of Colorado
Robin	Knox	134 Union Blvd Ste 665		Lakewood	CO	80228	303-236-4402	wnti.rknox@wispertel.net	Coordinator	Western Native Initiative
Carol	LoSapio	2150 Centre Avenue, Bldg A, Suite 300		Fort Collins	CO	80526	970-295-5731	closapio@fs.fed.us	Technical Publications Editor	USDA Forest Service, WOD, EMC, Publishing Arts
Nic	Medley	1201 Oakridge Drive, Suite 250		Fort Collins	CO	80525	970-225-3587	carl_medley@nps.gov	Fisheries Biologist	National Park Service
Kevin	Rogers	PO Box 775777		Steamboat Springs	CO	80477	970-846-7145	ke-vin.rogers@state.co.us	Aquatic Researcher	Colorado Division of Wildlife
Scott	Roth	P.O. Box 25486-DFC		Denver	CO	80225	303-236-4219	Scott_Roth@fws.gov	NFHAP/NFPP Coordinator	US Fish & Wildlife Service
John	Wullschleger	1201 Oakridge Drive, Suite 250		Fort Collins	CO	80525	970-225-3572	john_wullschleger@nps.gov	Fisheries Program Lead	National Park Service
Yoichiro	Kanno	1376 Storrs Road	308 Young Building	Storrs	CT	06269-4087	615-957-3925	yoichi-ro.kanno@uconn.edu	Doctoral candidate	University of Connecticut
Robert	Grieve	888 First Street N E		Washington	DC	20426	202-502-8752	ro-bert.grieve@ferc.gov	Fishery Biolo-gist	Federal Energy Regulato-ry Commission
Thomas	LoVullo	888 First Street N		Washington	DC	20426	202-502-8900	tho-	Fishery Biolo-	Federal Energy Regulato-

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
		E						mas.lovullo@ferc.gov	gist	ry Commission
John	Moore	1849 C St. NW (LSB 204)		Washington	DC	20240	202-912-7169	john_moore@blm.gov	Fish Biologist	Bureau of Land Management
David	Ayers	711 Lena Street		Salmon	ID	83467	(419) 631 - 1615	ayer9139@gmail.com	Fisheries Technician	Idaho Fish & Game
Arnie	Brimmer	1345 Barton Rd		Pocatello	ID	83204	208-251-0959	arnie.brimmer@idfg.idaho.gov	Regional Fishery Biologist	Idaho Department of Fish and Game
Jessica	Buelow	no address		Moscow	ID	83843	715-250-1500	buel0184@vandals.uidaho.edu	Student	University of Idaho
Tim	Copeland	1414 E Locust Ln		Nampa	ID	83686	208-465-8404	tim.copeland@idfg.idaho.gov	research biologist	Idaho Dept of Fish & Game
Jacob	Davis	no address		Moscow	ID	83843	218-340-7258	jacob.davis@sdstate.edu	Student	SD State University
Jim	DeRito	PO 550		Ashton	ID	83420	208-652-3567	jderito@henrysfork.org	Conservation Director	Henry's Fork Foundation
Bob	Esselman	99 Hwy 93 N.	P.O. Box 1336	Salmon	ID	83467	208-756-2271	bob.esselman@idfg.idaho.gov	Regional Fishery Biologist	Idaho Dept. of Fish and Game
Jon	Flinders	99 Hwy. 93 N.		Salmon	ID	83467	208-756-2271	jon.flinders@idfg.idaho.gov	Regional Fishery Biologist	Idaho Fish and Game Department
Mark	Gamblin	1345 Barton Rd.		Pocatello	ID	83204	208-232-4703	mark.gamblin@idfg.idaho.gov	Regional Supervisor	Idaho Department of Fish and Game
Kitty	Griswold	2410 Gail Drive		Pocatello	ID	83201	413-230-0405	griskitt@isu.edu	Wild Trout	Wild Trout
Amy	Haak	910 Main St	Suite 342	Boise	ID	83702	208-345-9800	ahaak@tu.org	Resource Information Director	Trout Unlimited
Brett	High	4279 Commerce Circle		Idaho Falls	ID	83401	208-525-7290	brett.high@idfg.idaho.gov	Regional Fisheries Biologist	Idaho Department of Fish and Game
Christine	Kozfkay	600 S. Walnut St		Boise	ID	83707	2089396713	christine.kozfkay@idfg.idaho.gov	Sr. Fisheries Biologist	IDFG
Elizabeth	Mamer	1414 E. Locust Lane		Nampa	ID	83686	208-465-8404 x270	liz.mamer@idfg.idaho.gov	Data Coordinator	Idaho Dept. of Fish & Game
Donald	Martin	1910 NW Blvd., Ste 208		Coeur D'Alene	ID	83814	208 665-0458	martin.don@epa.gov	Ecologist	US Environmental Protection Agency

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
Kevin	Meyer	1414 E. Locust Lane		Nampa	ID	83686	208-465-8404	kevin.meyer@idfg.idaho.gov	Fish Biologist	Idaho Fish and Game
Daniel	Schill	600 S. Walnut		Boise	ID	83712	208-465-8404	dan.schill@idfg.idaho.gov	Fishery Research Supervisor	Idaho Dept. of Fish & Game
Scott	Stanton	324 S 417 E Suite 1		Jerome	ID	83338	208 324 4359	scott.stanton@idfg.idaho.gov	Regional Fisheries Biologist	Idaho Fish and Game
Daniel	Steel	no address		Moscow	ID	83843	208-301-4188	stee7416@vandals.uidaho.edu	Student	University of Idaho
Kajsa	Stromberg	2110 Ironwood Pkwy		Coeur d'Alene	ID	83814	(208) 818-9928	kajsa.stromberg@deq.idaho.gov	Watershed Coordinator	Idaho Department of Environmental Quality
David	Teuscher	1345 Barton Road		Pocatello	ID	83204	208-232-4703	david.teuscher@idfg.idaho.gov	Regional Fisheries Manager	Idaho Department of Fish and Game
Charles	Warren	97 Hwy 93 North		Salmon	ID	83467	(208)756-6022	chuck.warren@idfg.idaho.gov	Regional Fisheries Biologist	Idaho Dept. of Fish and Game
Richard	Williams	524 W Two Rivers Drive		Eagle	ID	83616	208-938-9004	troutdna@cableone.net	Board Member	Fisheries Conservation Foundation
Laurence	Zuckerman	PO Box 1322		Salmon	ID	83467	208 756-8000	larry@westernwatersheds.org	Central Idaho Director	Western Watersheds Project
Joseph	McGurrin	401 Annette Ave		Stevensville	MD	21666	410-490-0570	jmcgurrin@tu.org	Resource Director	Trout Unlimited
Merry	Gallagher	Fisheries Research Section	650 State St	Bangor	ME	04401	207-941-4381	merry.gallagher@maine.gov	Fisheries Research Biologist	Maine Department of Inland Fisheries & Wildlife
Casey	Huckins	740 Dow Building	1400 Townsend Drive	Houghton	MI	49931	906-487-2475	cjhuckin@mtu.edu	Assoc. Professor	Michigan Technological University
Mark	Luttenton	Biology Department, Grand Valley State Univ.	1 Campus Drive	Allendale	MI	49401	616-331-2503	luttentm@gvsu.edu	Dr.	Grand Valley State University
Jeff	Lee	4700 West 77th Street		Minneapolis	MN	55435	952-832-2904	jefflee@barr.com	Senior Ecologist	Barr Engineering Company
Richard	Mayden	Dept. of Biology	3507 Laclede St.	St. Louis	MO	63103	314-814-0771	cypriniformes@gmail.com	Professor	Saint Louis University
Laurie	Battle	Department of	1300 W. Park	Butte	MT	59701	406-496-4857	LBattle@mtech.edu	Professor of	Montana Tech

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
		Mathematical Sciences, Montana Tech	St.						Mathematics	
Scott	Bosse	P.O. Box 1330		Bozeman	MT	59771	406-570-0455	sbosse@americanrivers.org	Northern Rockies Director	American Rivers
Dave	Bricker	PO Box 7		Ennis	MT	59729	406-682-3148	dbtwice@gmail.com	Director	Madison River Foundation
Scott	Christensen	PO Box 1874		Bozeman	MT	59771	406-586-1593	schristensen@greateryellowstone.org	Climate Change Program Director	Greater Yellowstone Coalition
Bob	Clark	PO Box 9283	210 N. Higgins, suite 222	Missoula	MT	59807	406-549-1142	bob.clark@sierraclub.org	Associate Regional Organizer	NGO - Sierra Club
Warren	Colyer	316 Burlington Ave.		Missoula	MT	59801	435-881-2149	wcolyer@tu.org	Watershed Programs Director	Trout Unlimited
Bruce	Farling	PO Box 7186	111 N. Higgins, Ste. 500	Missoula	MT	59807	406-543-0054	bruce@montanatu.org	Executive Director	Montana Trout Unlimited
Robert	Gresswell	2327 University Way, Suite 2		Bozeman	MT	59715	406-994-7085	bgresswell@usgs.gov	Research Biologist	USGS NoROCK
Kris	Homel	1805 S Rouse Ave		Bozeman	MT	59715	435.640.7026	kristen.homel@gmail.com	Fisheries Biologist	Montana Cooperative Fishery Research Unit
Travis	Horton	1400 S. 19th		Bozeman	MT	59718-5496	406-994-3155	thorton@mt.gov	FWP Region 3 Fisheries Manager	Montana Fish, Wildlife & Parks
Jacquelyn	Jones	108 North Star lane		Bozeman	MT	59718	406-587-0588	jjones@urbanifisheries.com	Project Coordinator	Urbani Fisheries, LLC
Joe	Kerkvliet	The Wilderness Society	503 West Mendenhall	Bozeman	MT	59715	406 581 9826	jkerkvliet@twsnw.org	Resource and Environmental Economist	The Wilderness Society
George	LaBar	6825 Hwy 91 North		Dillon	MT	59725	4066836671	glabar@3riversdbs.net	Dr.	Retired
Jason	Leppi	1302		S. 5th St. W.	MT	59801	406-579-0296	jleppi@yahoo.com	Research Assistant, NTSG	University of Montana
Richard	Lessner	PO Box 1527		Ennis	MT	59729	406-682-3148	mrfinfo@3rivers.net	Executive Director	Madison River Foundation

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
Mariah	Mayfield	Department of Ecology	310 Lewis Hall	Bozeman	MT	59717	541-905-9170	mariah.mayfield@gmail.com	Graduate Research Assistant	Montana State University
Thomas	McMahon	Ecology Department, Lewis Hall		Bozeman	MT	59717	406/994-2492	tmcmahon@montana.edu	Professor	Montana State University
joseph	naughton	720 highland park drive		missoula	MT	59803	406-493-0027	josef.naughton@gmail.com	graduate student	montana state university
Mark	Novak	Natural Resources Conservation Service	3550 Mullan Rd, suite 106	Missoula	MT	59808	406 829 3395 x119	mark.novak@mt.usda.gov	Biologist	NRCS
Bruce	Rieman	P.O. Box 1541		Seeley Lake	MT	59868	406-677-3813	brieman@fs.fed.us	Research Biologist Emeritus	Forest Service Rocky Mountain Research Station
Ted	Sedell	no address		Bozeman	MT	59717	406-224-1608	tsedell@gmail.com	Student	Montana State University
Adam	Sepulveda	1545 S 7th St. W		Missoula	MT	59801	406 243 2932	adam.sepulveda@mso.umt.edu	Dr.	University of Montana and USGS
peter	skidmore	p.o. box 1527		Bozeman	MT	59771	406-587-3949	restoringrivers@yahoo.com	Principal	Skidmore Restoration Consulting
Clinton	Smith	301 7th Ave. S.		Lewistown	MT	59457	307-752-3392	clinton.smith@msu.montana.edu	Graduate Research Assistant	MSU
james	thull	P.O.Box 173320		Bozeman	MT	59717	4069945305	jjthull@montana.edu	reference librarian	MSU
Joseph	Urbani	198 North Star Lane		Bozeman	MT	59718	406-581-8510	jurbani@urbanifisheries.com	Principal	Urbani Fisheries, LLC
Shane	Vatland	no address		Bozeman	MT	59717	406-579-6550	svatland@gmail.com	Student	Montana State University
Bob	Wiltshire	P.O. Box 1429		Livingston	MT	59047	406 222 7270	bob@stopans.org	Executive Director	Center for Aquatic Nuisance Species
Doug	Besler	645 Fish Hatchery Rd.		Marion	NC	28752	828-674-3278	doug.besler@ncwildlife.org	Regional Fishery Supervisor	North Carolina Wildlife Resources Commission
Jacob	Rash	645 Fish Hatchery Road		Marion	NC	28752	828-659-3324 x 225	jacob.rash@ncwildlife.org	Coldwater Research Coordinator	North Carolina Wildlife Resources Commission
LORIE	STROUP	Pisgah Ranger District	1001 Pisgah Highway	Pisgah Forest	NC	28768	(828)877-3350	loriestroup@fs.fed.us	Zone Fisheries Biologist	USDA Forest Service

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
James	MacCartney	18 Low Avenue, Suite 10		Concord	NH	03301	603-226-3436	jmaccartney@tu.org	Mr.	Trout Unlimited
Jason	Barnes	550 W. Plumb Ln	Suite B-237	Reno	NV	89509	775-232-9697	flyfishtruck-ee@yahoo.com	Member	Sagebrush Chapter, Trout Unlimited
Michael	Caltagirone	550 W. Plumb Ln	Suite B-237	Reno	NV	89509	775-232-9697	mcaltagi-rone@sagebrushtu.org	President	Sagebrush Chapter, Trout Unlimited
Chris	Crook-shanks	Nevada Depart-ment of Wildlife - Western Region	1100 Valley Road	Reno	NV	89512	775-688-1677	ccrook-shanks@ndow.org	Fisheries Biologist	Nevada Department of Wildlife
John	Elliott	Nevada Depart-ment of Wildlife	815 East 4th Street	Winnemucca	NV	89445	775-623-6565	jelliott@ndow.org	Fisheries Biologist	Nevada Department of Wildlife
Travis	Hawks	Nevada Depart-ment of Wildlife	815 East 4th Street	Winnemucca	NV	89445	775-623-6565	thawks@ndow.org	Fisheries Biologist	Nevada Department of Wildlife
Matt	Maples	Nevada Depart-ment of Wildlife - Western Region	1100 Valley Road	Reno	NV	89512	775-688-1677	mmaples@ndow.org	Fisheries Biologist	Nevada Department of Wildlife
Barbara	Knuth	501G Day Hall		Ithaca	NY	14853	607-255-5864	kjf29@cornell.edu	Vice Provost and Dean of Graduate School	Cornell University
Paul	Boehne	PO Box 907	1550 Dewey Ave	Baker City	OR	97814	5415231365	pboehne@fs.fed.us	Forest Fish Biologist	USDA Forest Service
John	Brinkley	2582 W 28th Avenue		Eugene	OR	97405	4083071332	mbrin-kle@comcst.net	Retired Fly Fisher	None
Erich	Gaedeke	805 S W Broad-way	Suite 550	Portland	OR	97205	503-552-2716	erich.gaedeke@ferc.gov	Fishery Biolo-gist	Federal Energy Regulato-ry Commission
Steve	Jacobs	28655 HWY 34		Corvallis	OR	97333	541 231 5801	steve.jacobs@oregonstate.edu	Research Fisheries Biologist	Oregon Department of Fish and Wildlife
Jimmy	Leal	1301 South G St.		Lakeview	OR	97630	(541) 947-6120	James_Leal@blm.gov	Fish Biologist	Bureau of Land Manage-ment
Cara	Rose	421 SW 6th Avenue, Suite 950		Portland	OR	97204	503-417-8700 x 6008	cara.rose@nfwf.org	Assistant Director	Jackson Hole One Fly - NFWF
Jim	Sedell	421 SW 6th Ave Ste 950		Portland	OR	97204	503-953-4323	jim.sedell@nfwf.org	Director Fish Conservation	National Fish and Wildlife Foundation
Jack	Williams	329 Crater Lake Avenue		Medford	OR	97504	541-261-3960	jwilliams@tu.org	Chief Scientist	Trout Unlimited
Robert	Carline	123 Gibson Place		Port Matilda	PA	16870	8142383119	m-bcar-	Retired	Retired

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
								line@comcast.net		
Jason	Detar	450 Robinson Lane		Bellefonte	PA	16823	814-359-5118	jdetar@state.pa.us	Fisheries Biologist	Pennsylvania Fish and Boat Commission
Jefferson	Deweber	241 W Whitehall Rd		State College	PA	16801	901-387-7102	jtdeweber@gmail.com	PhD Student	Penn State
William	Sharpe	126 Land and Water Bldg.		University Park	PA	16802	814-863-8564	wes@psu.edu	Professor Emeritus	Penn State University
Amy	Wolfe	18 E. Main St, Suite 3		Lock Haven	PA	17745	570-748-4901	awolfe@tu.org	Eastern Abandoned Mine Program Director	Trout Unlimited
Daniel	James	SDSU - SNP 138		Brookings	SD	57007	6057164484	Daniel.James@sdstate.edu	Research Assistant (Ph.D.)	South Dakota State University
Ron	Koth	122 S. Berry Pine Rd		Rapid City	SD	57702	6053901065	ron@rushmore.com	Senior Fisheries Biologist	Barr Engineering Co.
Jerry	Wilhite	3305 West South St.		Rapid City	SD	57702	605-394-2391	jerry.wilhite@state.sd.us	Fisheries Biologist	South Dakota Game, Fish and Parks
Stephen	Moore	Great Smoky Mountains National Park	107 Park Headquarters Road	Gatlinburg	TN	37738	865 - 436 - 1250	steve_e_moore@nps.gov	SupervisoryFishery Biologist	USDI - National Park Service
Tracy	Bowerman	no address		Logan	UT	84321	541-915-3910	tracybowerman@gmail.com	Student	Utah State University
Paul	Evans	571 WIDB		Provo	UT	84602	801.422.3259	evansp@byu.edu	Dr.	Brigham Young University
Dennis K.	Shiozawa	Department of Biology	147-A WIDB	Provo	UT	84602	801-422-4972	shiozawa@byu.edu	Professor	Brigham Young University
Ken	Theis	535 E 380 S		Smithfield	UT	84335	435-881-4336	kdtheis@gmail.com	Graduate Student	Utah State University
Andy	Dolloff	1710 Ramble Rd		Blacksburg	VA	24060	540 231-0069	adoll@vt.edu	Fishery Scientist	USDA Forest Service
Alexandra	Fitzgerald	10783 Cannon Hill Drive		Broadway	VA	22815	540-255-5999	bradly.trumbo@gmail.com	Graduate Assistant	FS/LSU
Anna	Harris	4401 N Fairfax Dr, MS-4020		Arlington	VA	22203	7033581842	anna_harris@fws.gov	Economist	US Fish & Wildlife Service
Mark	Hudy	701 Carrier Drive, MSC 4114		Harrisonburg	VA	22807	540-568-3508	hudymx@jmu.edu	National Aquatic Ecologist	US Forest Service

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
Robert	Humston	Department of Biology, Washington & Lee University	301 Howe Hall	Lexington	VA	24450	540-458-8341	humstonr@wlu.edu	Assistant Professor of Biology	Washington & Lee University
Chastine	Kyger	820 Madison Drive	MSC 7801	Harrisonburg	VA	22807	540-250-3939	cdkyger@gmail.com	Graduate Assistant	USFS/JMU
Edward	Maillett	4401 N. Fairfax Dr.	MS 7081-43	Arlington	VA	22203	703-358-2322	edward_maillett@fws.gov	Economist	U.S. Fish and Wildlife Service
Steve	McMullin	Department of Fisheries & Wildlife Sciences	108 Cheatham Hall	Blacksburg	VA	24061	540-231-8847	smcmulli@vt.edu	Associate Department Head	Virginia Tech
Craig	Roghair	1710 Ramble Rd		Blacksburg	VA	24060	540 231-0078	croghair@fs.fed.us	Fishery Biologist	USDA Forest Service
eric	smith	Virginia Tech		Blacksburg	VA	24060	5402315657	epsmith@vt.edu	Prof	Statistics Department
Bradly	Trumbo	820 Madison Drive	MSC 7801	Harrisonburg	VA	22807	540-246-2598	trumboba@dukes.jmu.edu	Graduate Assistant	USFS/JMU
William	Grover	1 Winooski Park		Colchester	VT	05439	802-654-1463	wgrover@smcvt.edu	Political Science Professor	Saint Michael's College
Jennifer	Brown	375 W. Main St.	Apt. 3	Pullman	WA	99163	(406) 925-0915	jen_brown@wsu.edu	Ph.D. Candidate	Washington State University
Patrick	Connolly	5501A Cook Underwood Road		Cook	WA	98605	509-538-2299x269	pconnolly@usgs.gov	Lead Research Fish Biologist	USGS Columbia River Research Lab
Maureen	Kavanagh	1211 SE Cardinal Ct. Suite 100		Vancouver	WA	98683	360-604-2500	maureen_kavanagh@fws.gov	Fish Biologist	USFWS-Columbia River Fisheries Program Office
Miranda	Plumb	510 Desmond Drive SE, Suite 102		Lacey	WA	98503	360.753.9440	miranda_plumb@fws.gov	Fish and Wildlife Biologist	U.S. Fish and Wildlife Service
Ray	White	320 12th Ave. N.		Edmonds	WA	98020-2930	425-672-8268	rw@seanet.com	Dr.	Montana State Univ. (Retired)
Ben	Kissinger	1895 Lea Rd		Mosinee	WI	54455	715-551-9284	bkiss024@uwsp.edu	Lab Technician	Wisconsin Cooperative Fisheries Research Unit
Matthew	Mitro	2801 Progress Rd.	WDNR Science Operations Center	Madison	WI	53716	608-221-6366	matthew.mitro@wisconsin.gov	Coldwater Fisheries Research Scientist	Wisconsin DNR

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
Pete	Segerson	910 Hwy 54 East		Black River Falls	WI	54615	715 284 1447	pe-ter.segerson@wi.gov	Field Operations Supervisor	WI Dept. Natural Resources
David	Seibel	223 East Steinfest Ave		Antigo	WI	54409	715-623-4190	da-vid.seibel@wisconsin.gov	Fisheries Biologist	Wisconsin Department of Natural Resources
Richard	Stewart`	3911 Fish Hatchery Road		Fitchburg	WI	53711	(608) 273-5967	Ri-chard.Stewart@wisconsin.gov	Regional Fisheries Supervisor	Wisconsin Department of Natural Resources
Walter	Brown	PO Box 67		Elkins	WV	26241	3045311365	steve-brown@wvdnr.gov	Supervisor, Stream Restoration	WV Division of Natural Resources
Timothy	King	Leetown Science Center	11649 Leetown Road	Kearneysville	WV	25430	304-724-4450	tlking@usgs.gov	Dr.	USGS
Michael	Shingleton	PO Box 67		Elkins	WV	26241	304-637-0245	mikeshingle-ton@wvdnr.gov	Assist. Chief-Coldwater Fish Management	WV Division of Natural Resources (WVDNR)
Pat	Bigelow	PO Box 473		Yellowstone National Park	WY	82190	406-848-9805	stepha-nie.stratton@montana.edu	Wild Trout	Yellowstone National Park
Michael	Bower	2013 Eastside 2nd Street		Sheridan	WY	82801	307-674-2621	mbower@fs.fed.us	Fishery Biologist	Bighorn National Forest
Kathy	Buchner	PO Box 1022		Jackson	WY	83001	307-733-4944	kbuch-ner@wyoming.com	Treasurer	Wyoming Trout Unlimited
Jason	Burckhardt	2820 State Highway 120		Cody	WY	82414	307-527-7125	ja-son.burckhardt@wgf.state.wy.us	Fisheries Biologist	Wyoming Game and Fish Department
Brendan	Crowley	PO Box 220		Pinedale	WY	82941	307-367-4326	jrneal@fs.fed.us	Fisheries	Bridger-Teton National Forest
Lance	Downing	170 North First Street		Lander	WY	82520	307-332-2159	lance_downing@fws.gov	Technician	US Fish and Wildlife Service
Steve	Gale	528 South Adams Street		Laramie	WY	82070	307-745-4046	Steve.Gale@wgf.state.wy.us	Fisheries Biologist	Wyoming Game and Fish Department
Thomas	Gardiner	PO Box 220		Pinedale	WY	82941	307-367-4326	tagardin-er@fs.fed.us	Fisheries Biologist	Bridger-Teton National Forest
Rob	Gipson	P.O. Box 67		Jackson	WY	83001	(307)733-2321	rob.gipson@wgf.state.wy.us	Regional Fisheries Supervisor	WY Game and Fish
Stephen	Gray	Dept. 3943	1000 E. University Ave	Laramie	WY	82072	307-766-6659	sgray8@uwyo.edu	Wyoming State Clima-	University of Wyoming

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
									tologist	
Greg	Hill	PO Box 220		Pinedale	WY	82941	307-367-4326	jrneal@fs.fed.us	Fisheries	Bridger-Teton National Forest
Pat	Hnilicka	170 North First Street		Lander	WY	82520	307-332-2159	pat_hnilicka@fws.gov	Fish and Wildlife Biologist	US Fish and Wildlife Service
Kevin	Johnson	260 Buena Vista Drive		Landerq	WY	82520	307.332.2688	kevin.johnson@wgf.state.wy.us	Regional Fisheries Supervisor	Wyoming Game and Fish Department
Adam	LaDell	PO Box 220		Pinedale	WY	82941	307-367-4326	jrneal@fs.fed.us	Fisheries	Bridger-Teton National Forest
Matt	Lubejko	PO Box 220		Pinedale	WY	82941	307-367-4326	jrneal@fs.fed.us	Fisheries	Bridger-Teton National Forest
Dan	Mahony	PO Box 636		Yellowstone National Park	WY	82190	307-344-2280	dan_mahoney@nps.gov	Wild Trout	National Park Service
Leland	McDonald	528 S Adams Street		Laramie	WY	82070	307-745-4046	leland.mcdonald@wgf.state.wy.us	Fisheries Biologist	Wyoming Game and Fish Department
Dirk	Miller	5400 Bishop Boulevard		Cheyenne	WY	82006	307-274-2812	dirk.miller@wgf.state.wy.us	Fisheries Management Coordinator	Wyoming Game and Fish Department
Dan	Moede	3611 Concord Rd		Cheyenne	WY	82601	307-634-1042	dmoede@bresnan.net	Wild Trout	Wild Trout
Andrew	Morin	PO Box 220		Pinedale	WY	82941	307-367-4326	jrneal@fs.fed.us	Fisheries	Bridger-Teton National Forest
Joe	Neal	PO Box 220		Pinedale	WY	82941	307-367-5730	jrneal@fs.fed.us	Fisheries Biologist	Bridger-Teton National Forest
Cathy	Purves	250 North 1st Street	PO Box 64	Lander	WY	82520	307-332-6700	cpurves@tu.org	Science & Technical Advisor	Trout Unlimited
Dave	Skates	170 North First Street		Lander	WY	82520	307-332-2159	david_skates@fws.gov	Project Leader	US Fish and Wildlife Service
Mark	Smith	2820 Hwy 120		Cody	WY	82414	307-899-3122	mark.smith@wgf.state.wy.us	Fisheries Biologist	Wyoming Game and Fish
Mike	Snigg	528 South Adams		Laramie	WY	82070	1-307-745-4046	mike.snigg@wgf.state.wy.us	Laramie Region Fisheries Supervisor	Wyoming Game and Fish Department
Tracy	Stephens	PO Box 67		Jackson	WY	83001	3077332321	Tracy.Stephens@wgf.s	Fisheries Biologist	WY Game and Fish Dept

First Name	Last Name	Street 1	Street 2	City	State	Zip	Phone	Email	Title	Agency
								tate.wy.us		
Jeff	Streeter	250 N 1st		Lander	WY	82520	307 332 7700	jstreeter@tu.org	North Platte Project Manager	Trout Unlimited
Cory	Toye	250 N 1st		Lander	WY	82520	307 332 7700	ctoye@tu.org	Wyoming Project Manager	Trout Unlimited
Nick	Walrath	250 N 1st		Lander	WY	82520	307 332 7700	njwalrath@gmail.com	Upper Green River Project Manager	Trout Unlimited
Chad	Whaley	P. O. Box 170		Moose	WY	83012	307-739-3666	chad_whaley@nps.gov	Fisheries Technician	Grand Teton National Park
Steve	Yekel	2820 State Hwy 120		CODY	WY	82414	307-527-7125	steve.yekel@wgf.state.wy.us	Fisheries Supervisor	Wyoming Game and Fish Dept.
Dave	Zafft	528 S. Adams St.		Laramie	WY	82070	307-745-5180 x235	david.zafft@wgf.state.wy.us	Assistant Fish Mgmt Coordinator	Wyoming Game and Fish Department
Ray	Zubik	808 Meadow Lane		Cody	WY	82414	(307) 578-5160	rzubik@fs.fed.us	Fisheries Biologist	Shoshone National Forest

Document design and publishing by USDA Forest Service, Ecosystem Management Coordination, WOD, Publishing Arts, Carol LoSapio, December 2010.

Save the Date

A Reminder . . . WTXI will be held in 2014.
Stay in contact through: www.wildtroutsymposium.com

Wild Trout XI

2014