

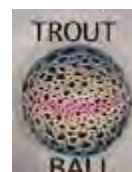
Science, Politics, and Wild Trout Management:

Who's Driving and Where are We Going?



Proceedings of the Wild Trout XII Symposium

Holiday Inn, West Yellowstone, Montana
September 26-29, 2017



Wild Trout XII

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Proceedings of the Wild Trout XII Symposium

West Yellowstone, MT
September 26-29, 2017

Symposium Chair: Bob Gresswell
Proceedings Editor: Bob Carline
Program and Proceedings Designer: Carol LoSapio

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*A special thanks to the U.S. Forest Service for supporting the Wild Trout Symposium's technical transfer of information related to wild trout research and management and with the costs for symposium proceedings and other printed materials.



A Reminder . . . WTXIII will be held in 2020.

Stay in contact through:

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Thank you to Eric Stark, our wonderful photographer, who let us use his photos for this publication.

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Wild Trout XII Symposium Program





Program Schedule

Tuesday, September 26th, 2017

- 2:00 Big Sky Anglers casting demo
- 4:00 Wild Trout XII Steering Committee Meeting
Volunteer Meeting / AV Table Open
- 5:00 Registration Opens
Speaker Presentation Check-in Vendor / Poster
set up
- 6:30 Welcome Social

Wednesday, September 27th, 2017

- 6:30 Continental breakfast
- 7:00 Registration Opens
Speaker Presentation Check-in Vendor / Poster
set up
- 8:00 Welcome and Plenary Sessions
- 10:10 Break
- 10:40 Session 1: Anglers, Stakeholders, and the
Socioeconomics of Wild Trout
- 12:00 Lunch with Dan Wenk, YNP Superintendent
- 1:30 Session 2: Threats and Management of Stream
Habitat: A Look Into the Future
- 3:10 Break
- 3:40 Session 3: Conservation Genetics and Genomics of
Coldwater Fishes: A Tribute to Dr. Tim King
- 5:00 Poster Session I
- 6:30 Dinner

Thursday, September 28th, 2017

- 6:30 Continental breakfast
- 8:00 Session 4: Population Dynamics and Ecology of
Wild Trout
- 9:40 Break
- 10:10 Session 4: Population Dynamics and Ecology of
Wild Trout (continued)
- 10:30 Session 5: Native Trout Conservation
- 11:50 Awards Luncheon
- 1:00 Session 5: Native Trout Conservation (continued)
- 2:40 Break
- 3:10 Session 6: Nonnative Fishes and Tools for Native
Trout Management
- 5:00 Poster Session II
- 6:30 Dinner

Friday, September 29th, 2017

- 6:30 Continental breakfast
- 8:00 Martha Williams, Director MT FWP
- 8:20 Session 7: Stream Habitat Management: Traditional
and New Approaches
- 10:00 Break
- 10:30 Session 8: Disease, Parasites, and the Health of
Wild Trout: Should We Be Concerned?
- 12:10 Closing Remarks
- 12:20 Adjourn
- 12:30 Post-symposium debrief and WTXIII Planning
(box lunch provided if you participate!
All welcome to attend.)



EVENT 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



Trout Unlimited

Today Trout Unlimited (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.



Flycasters, Inc. of San Jose

Flycasters, Inc. of San Jose is proud to be a sponsor of Wild Trout Symposium XII in honor of the late Marty Seldon. Flycasters was honored to have Marty Seldon as a member who was very active in fisheries conservation for our club, the Federation of Fly Fishers (FFF) and on the organizing committee of every WT Symposium until 2010. Marty said that, "I'm a firm believer of catch-and-release and very concerned about the preservation of the genetics of our wild fish for future generations." Marty received the first Aldo Starker Leopold Award at the WT Symposium in 1984. The Flycasters is very proud of Marty and for his lifetime of service to preserving wild trout. The Flycasters of San Jose is a large fly fishing club promoting and teaching all aspects of fly fishing with a history of supporting fisheries conservation and the preservation of wild trout.



Idaho Department of Fish and Game

The mission of the Idaho Department of Fish and Game (IDFG) is that all wildlife, including wild animals, wild birds, and fish, within the state of Idaho, is hereby declared to be the property of the state of Idaho. It shall be preserved, protected, perpetuated, and managed. It shall only be captured or taken at such times or places, under such conditions, or by such means, or in such manner, as will preserve, protect, and perpetuate such wildlife, and provide for the citizens of this state and, as by law permitted to others, continued supplies of such wildlife for hunting, fishing and trapping.

Supporting Sponsors

Henry's Fork Foundation



For nearly 30 years, the Henry's Fork Foundation has been the only organization whose sole purpose is to conserve, protect, and restore the unique fisheries, wildlife, and aesthetic qualities of the Henry's Fork of the Snake River and its watershed. The Foundation is an advocate for wild trout, and its credibility has earned it a seat at the table in managing water resources to benefit the fishery and those who depend on it, local businesses that rely on the health of the river, and anglers from all over the world who love the river and help fuel the local economy.



McKenzie Flyfishers

The McKenzie Flyfishers are comprised of a group of people who share a common interest in flyfishing. The Club, based in Eugene, Oregon, was conceived and organized in April of 1964 to: [1] Enjoy social contact with others interested in fly fishing; [2] Encourage fly fishing as a method of angling; and [3] Protect and

increase the fishery resources. The club is active in teaching the craft of flyfishing at a variety of levels, engaging in numerous fishing outings throughout the Pacific Northwest and our members fish throughout the world. In addition, we participate in numerous conservation activities, including Citizen Science studies, cleanup, and restoration in and around our home waters and advocate for the preservation of our fisheries through policy and legislative opportunities.



**WESTERN
NATIVE
TROUT
INITIATIVE**

Western Native Trout Initiative

The Western Native Trout Initiative (WNTI), established in 2006, is a public-private Fish Habitat Partnership that works collaboratively across 12 western states to conserve, protect, restore and recover 21 native trout and char species. Covering over 1.75 million square miles of public and privately managed lands, WNTI and

its partners combine science-based assessments with expert and local knowledge to establish joint priorities for native trout conservation at a landscape scale.



Wyoming Game & Fish Department

Wyoming's wealth of fish and wildlife is the result of sportsmen and women who have paid for its management through licenses fees since the creation of the Game and Fish Commission inception in 1921.

As with many wildlife agencies, hunters and anglers have traditionally provided nearly all the financial resources to support wildlife management, with 80% of our funds coming from license fees and excise taxes on hunting and fishing equipment. Only about 6% of our funding comes from the State's General fund which is used for specific programs: Aquatic Invasive Species, Sage Grouse, Veterinary Services, and wolf management. The additional funds come from a variety of sources including stamps, fees and various grants.

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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.

Beyond Words

Beyond Words

A writing, editing, and graphic design service, Carol LoSapio (owner) of Beyond Words, provides multimedia communication products (strategic plans, technical guides, slide shows, symposium proceedings, posters, etc.). "We specialize in finding the right words and the right look to reach your targeted audience." We have a proven track record for success and have been recognized by the Society for Technical Communicators and the National Association of Government Communicators for outstanding presentation of technical information. Our goal is to use "plain language" and visuals to enhance and produce YOUR final product. An avid fly fisher, Carol has attended and produced the symposium proceedings for Wild Trout, Inc. since 1997 and enjoys supporting wild trout management and sustainability.



Big Sky Anglers

Based out of West Yellowstone, MT, Big Sky Anglers is a full service fly shop, outfitter, and angling travel company specializing in equipping, guiding, and teaching anglers of all ages and skill levels.

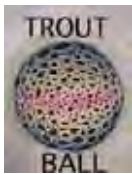
They offer guided fly-fishing throughout the region on famed waters of southwest Montana, Yellowstone National Park, and the Henry's Fork of the Snake River and Henry's Lake in Idaho.



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.



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.



Simms

Simms is a fishing company.

Founded on the pillars of innovation we strive to build the highest quality products to keep anglers dry, comfortable, and protected from the elements - no matter the conditions. The company was the brainchild of visionary angler John Simms who saw a need to develop better waders and accessories than what was then available on the market. That quest led to the development of Simms Fishing Products in 1980. During that era, Simms was one of the first companies worldwide to introduce neoprene waders, which provided enhanced warmth and waterproofing gear for serious anglers pushing the limits of their fishing pursuits.

Today, Simms continues to take the fishing market by storm with a trained eye on fisheries conservation and

inspired product development of the world's premier technical fishing apparel, footwear, and equipment.



Smith Root

Since 1964, Smith-Root has proudly partnered with fisheries scientists to develop solutions for the fisheries conservation community. Smith-Root

is proud to be a strong supporter of the Wild Trout Symposium.



U.S. Forest Service

Wild Trout would like to extend a special thanks to the U.S. Forest Service for supporting the Wild

Trout Symposium's technical transfer of information related to wild trout research and management by helping with costs for symposium proceedings and other printed materials.

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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.

WILD TROUT XII AWARDS

Awards Committee:
Andy Dolloff, Bob Carline, and
Bob Gresswell

Awards Selection

The Awards Committee is chosen from the membership of the Organizing Committee. Several months prior to the symposium, the committee posts requests for nominations along with applications and instructions on the symposium website. The following awards were made at Wild Trout XII in 2017.

The applicant pool is solicited from university faculties and representatives of AFS student chapters, Trout Unlimited members and chapters, and a broad cross-section of scientists and managers.

Awards have been created over the years by the organizing committees of the Wild Trout Symposium in honor of individuals who have made outstanding contributions toward the advancement and stewardship of wild trout. Notifications of these awards occur in the summer prior to the meeting and recipients are encouraged to attend the symposium to personally receive their award during the Awards Banquet. The following awards were made at Wild Trout XII in 2017.

Aldo Starker Leopold Wild Trout Award

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

Aldo Starker Leopold Wild Trout Award – Professional

The Aldo Starker Leopold Wild Trout Award in the Professional category is presented to an individual for outstanding contributions to the protection and conservation of wild trout resources.

A qualified nominee is or has been significantly involved in any or all of the following areas of wild trout management or conservation:

- ▶ Educational activities (mentoring, etc.)
- ▶ Raising awareness of wild trout issues and conservation
- ▶ Support for agency conservation activities (e.g. fund raising, volunteering time, etc.)

Congratulations Joe McGurrin

Joe has devoted his career to conserving wild trout across North America. He has spent more than 25 years with Trout Unlimited and has been instrumental in developing TU's conservation plans for wild trout on Federal lands and on state-owned lands. He has been a leader in formulating TU's "Home Rivers Program", an enormously successful venture that has expanded to 12 major restoration projects across the U.S. Over and over, Joe has demonstrated his passion for native trout conservation and restoration. He has been particularly effective in bringing together Federal, state, tribal, and local organizations to support native trout conservation. He spearheaded



efforts to raise funds for Gila Trout restoration in Arizona and New Mexico, where great strides have been made in reintroducing Gila Trout to burned over watersheds. His passion and strategic vision for native trout conservation has helped TU leverage a myriad of funding sources, build new partnerships, recruit volunteer boots on the ground, and apply science to direct on-the-ground conservation work.

Aldo Starker Leopold Wild Trout Medal - Non-Professional

The Aldo Starker Leopold Wild Trout Award in the Non-Professional category is presented to an individual for outstanding contributions to the protection and conservation of wild trout resources.

A qualified nominee would be avidly involved in any or all of the following aspects of wild trout conservation:

- ▶ Educational activities (mentoring, etc.)
- ▶ Raising awareness of wild trout issues and conservation
- ▶ Support for agency conservation activities (e.g. fund raising, volunteering time, etc.)

Congratulations David Sweet

David Sweet has been a champion for conservation of the Yellowstone Cutthroat Trout. He has made countless presentations to Trout Unlimited Chapters and other public audiences throughout Wyoming,

Colorado, and Montana, educating them about the importance of supporting efforts to save Cutthroat Trout, particularly in Yellowstone Lake. He has been instrumental in raising hundreds of thousands of dollars to pay for control of Lake Trout, which had devastated Yellowstone Cutthroat Trout in Yellowstone Lake. He has organized several volunteer projects to assist in Lake Trout suppression efforts, tag Lake Trout, and conduct movement studies. His enormous efforts have greatly enhanced the National Park Service's successful program to suppress Lake Trout and assist the recovery of Yellowstone Cutthroat Trout.

Marty Seldon Graduate Student Scholarship Awards

Marty loved wild trout and the places they lived. He continually challenged the organizing committees for each symposium to find the best research and management biologist internationally to come and present the results their work.

Marty was also believed that students held the keys to the future and needed to play significant roles in the symposium.

As the Organizing Committee developed plans for Wild Trout IX, they discussed ways to recognize



Aldo Starker Leopold Award Winners Dave Sweet and Joe McGumin

Marty's 30+ years of involvement and many contributions to the Symposium. The result was the Marty Seldon Student Scholarship Award. Two awards are to be presented at each symposium.

The award is intended to support outstanding students who conduct research on coldwater fisheries ecology and management. The award, initially established at \$500, is intended to assist recipients with travel or other costs associated with attending the Wild Trout Symposium.

WTXII Marty Seldon Graduate Student Scholarship

Congratulations Shannon White



Shannon is a PhD Candidate in the Ecology Program at Pennsylvania State University. Her dissertation merges individual, population, and species ecology to gain a better understanding of population persistence and adaptive potential in native Brook Trout. Her work seeks to understand the evolutionary significance

of individual variation and how genetic and behavioral diversity increases population resiliency. Shannon is an active member in the American Fisheries Society and maintains a popular science outreach website to engage anglers and citizen scientists with topical issues in coldwater fish conservation. She hopes to settle into a university teaching position where she can continue pursuing novel research while engaging the next generation of fisheries scientists.

WTXII Marty Seldon Graduate Student Scholarship

Congratulations Travis Rehm



Travis is a Graduate Research Assistant at South Dakota State University. He is currently studying Brown Trout ecology in the Black Hills of South Dakota.

Before beginning his masters degree program, Travis worked in fisheries for a variety of state, federal, and private agencies,

including Montana, Idaho, US Forest Service and US Geological Survey.

He grew up in Southwest Montana and developed a love affair with the local trout at a very early age. He lives and breathes all things trout, in particular native trout, and hopes to continue working with them upon completion of his graduate degree.

WTXII Marty Seldon Graduate Student Scholarship

Congratulations Olivia Sparrow



Olivia is pursuing her MS in Civil Engineering at the University of Minnesota and is a practicing Water Resources Engineer at Emmons & Olivier Resources, Inc. with clients across Canada and the Mid-West.

She has devoted her career to restoring water quality and ecosystem health in urban and rural watersheds.

Her research and work interests include developing tools for watershed management decision making as well as establishing best management practices and urban development standards for improving water quality.

Olivia's graduate research compares the shade provided by grassy and woody riparian vegetation as controls for baseflow stream temperature in a designated trout stream in Stillwater, Minnesota.

Ron Remmick Undergraduate Student Scholarship Award

The Ron Remmick Undergraduate Student Scholarship Award was established at WT IX to memorialize Ron Remmick's 25-year career with Wyoming Game and Fish. A recognized authority on Colorado River and Bonneville cutthroat trout management, Ron was at the cutting edge of native fish conservation and restoration, initiation of joint agency programs, and leader in public education. His knowledge was prodigious, his approach innovative, and his enthusiasm contagious.

Ron was a true champion of native fish restoration and conservation, admired and respected by his fishy

colleagues, the fellow employees of Wyoming Game and Fish, and the American Fisheries Society.

This award recognizes one outstanding undergraduate student who has demonstrated an interest in conservation and restoration of native trout – and encourages them to attend the Wild Trout Symposium.

The award includes a \$400 stipend to assist student travel or other costs incurred in attendance of this symposium. The Ron Remmick Scholarship is open to undergraduate students in Fisheries Management or related fields.

Congratulations Ben Martin

Ben Martin was selected as the awardee for the Ron Remmick Scholarship. He is a senior at Juniata

College and is pursuing a BS in Environmental Science. His focus is on aquatic ecology.

Ben's seasonal field experience includes impacts of natural gas development on headwater streams, stable isotope analysis, headwater food webs, and morphological assessment of largemouth bass in Maine and Pennsylvania

His near-term career goal is graduate school and ultimately a PhD in fisheries ecology or a closely related field.

Ben is a coauthor with Dr. Christopher Grant on an article titled: "Fracked fish? Effects of Unconventional Natural Gas Extraction on wild brook trout (*Salvelinus fontinalis*)".



Ben Martin and Andy Dolloff



Dave Sweet and Andy Dolloff



Joe McGurrin and Andy Dolloff



Shannon White and Andy Dolloff



Olivia Sparrow and Andy Dolloff



Travis Rehm and Andy Dolloff

Plenary Session





Driftless Waters: A Tale of Destruction, Renewal and Hope for the Future

David M. Vetrano

Fisheries Biologist, Wisconsin Department of Natural Resources (Retired)

The Driftless Area of SW Wisconsin, SE Minnesota, NE Iowa, and NW Illinois is a unique landform of the United States. There is no evidence that the last glaciation altered the area unlike most of North America. This lack of glacial drift gave the Driftless Area its name. Unfortunately, there has been an enormous amount of change in the landscape since Europeans settled the area.

Although there had been travellers through the Driftless Area since the late 1500's, it wasn't until the 1820's that the major migration of mostly northern Europeans occurred. They found a landscape that looks significantly different than it does now. Most of the land on either side of the Mississippi River was tall grass prairie or oak savannah. The predominant landforms are coulees, from the French verb "couler" which means, "to flow". Limestone and sandstone bluffs that tower some 400 feet above the valley floor characterize it. The first settlers found a plethora of narrow, deep, crystal clear, spring fed streams that were full of Brook Trout *Salvelinus fontinalis*. Records of 18- to 20-inch fish were not uncommon.

Logging was the first industry with dozens of sawmill sites using the abundant water resources to float millions of board feet of logs from the great forests to the north. Agriculture did not become a major industry until the 1850's with advent of the moldboard plow that was able to cut through the thick sod layers of the prairies. The first crop was wheat as this was the grain early farmers were most familiar with. Wheat was "king" until the 1880's when dairy became the main industry and remains the main industry today.

Unfortunately the "up and down" farming practices that worked well in northern Europe where precipitation may only be 10 inches per year were unsuited to a climate with 32 inches of annual precipitation. In addition, the "loess" soils of the region have a consistency of melted ice cream when they are saturated. Hillside dairy grazing quickly denuded the vegetation and the animals' hooves compacted the soil preventing percolation of rainwater and snowmelt. Aldo Leopold later referred to this

phenomenon as "water off a tin roof". Soon "rills" began to form. These became head cuts, then gullies, then small canyons. Flash flooding which was rare before European settlement became common by the early 1900's as millions of tons of sediment started their downslope movement. By the 1930's some 12 to 15 feet of sediment had eroded off the hillsides into all of the valley floors on both sides of the Mississippi River. Accretion rates were 2 to 3 inches each year. The Kickapoo watershed in SW Wisconsin alone had 36,000 acre-feet of sediment that had eroded into the valley. If this soil were placed on a NFL playing field the result would be a "dirt monument" reaching 12.4 miles into the sky. As sediment inundated the valleys roads, bridges and fences had to be rebuilt as the earlier ones were buried by tons of soil.

Not surprising, the Brook Trout fishery also suffered. Lower stream sections became deeply entrenched and middle and upper reaches lost their defined channel and became braided. Instream habitat was lost. Spring flow and base flow were reduced as surface water runoff exceeded groundwater recharge. As streams became wide, shallow and unstable, water temperatures rose and the Brook Trout fishery was replaced by species more associated with warm water.

To their credit, these farmers realized early that the massive amount of soil erosion occurring was the limiting factor to economic stability in the Driftless Area. They petitioned the federal government for help. This resulted in the nation's first watershed project just outside of Coon Valley, Wisconsin. At an experimental farm, the Soil Erosion Service was formed. This later became the Soil Conservation Service and is now the Natural Resource Conservation Service. At this site farming practices that are now standard in the Driftless Area (contour strips, terraces, grass waterways, etc.) were developed and perfected.

By this time Brown Trout *Salmo trutta* were stocked in area streams, as they are more tolerant of the warmer, more turbid stream conditions. Postwar rod and gun clubs initiated some habitat restoration efforts in the 1950's to provide overhead cover for the put-and-take fishery. By the 1970's, some stream conditions were improving as better farming practices allowed more groundwater infiltration to occur.

Although some carryover of stocked Brown Trout occurred, little or no natural reproduction could be found in most waters, as stream temperatures remained high.

Instream habitat structures were short lived as little attention was given to reconnecting the stream to its floodplain, allowing the still frequent flash floods to erode around the single wing deflectors commonly used leaving them high and dry. In the early 1980's Wisconsin DNR made major change in instream habitat efforts in the Driftless Area by developing a different overhead structure (LUNKERS) and by sloping the stream banks to reconnect the stream to its floodplain. As a result, floods no longer caused the amount of damage that was common with earlier efforts.

The 1985 Farm Bill proved to be a watershed event (pun intended) resulting in more groundwater percolation. The Conservation Reserve Program (CRP) paid farmers to idle and plant perennial vegetation on thousands of acres. Cross Compliance required producers receiving any agricultural subsidies to have and follow a conservation tillage plan on their farms. By the late 1980's, base flow and spring flow increased as more perennial vegetation improved groundwater infiltration resulting in colder stream temperatures. Fisheries surveys in many streams found more carryover of Brown Trout and for the first time natural reproduction as stream conditions improved.

Local efforts by fisheries personnel to improve trout survival resulted in an experimental stocking program of "feral" Brown Trout and Brook Trout. Adults from naturally reproducing, non-stocked streams were stripped of eggs and milt and the subsequent young were raised in a partially covered raceway with automatic feeders to keep human contact to a minimum. To compare survival of the feral fish against the hatchery strains, matched cohorts were stocked in several streams. A year later the feral trout had out survived the domestic strain fish by a factor of 6:1. A statewide wild trout program was initiated in 1995.

By this time the number of non-local anglers (driving more than 50 miles) increased significantly as word of the ever-improving fishery in Driftless Area spread. Entrepreneurs catered to more urban anglers by providing lodging and more upscale dining

experiences. By 2008, a Trout Unlimited economic study found that trout fishing in the entire Driftless Area was a 1.1 billion dollar industry and growing.

Unfortunately, some of the same issues that plagued the streams in the 1930's still exist. When commodity prices reached record levels several years ago much of the long idled or conservation tillage acreage was plowed up and planted into row crops. "Up and down" farming increased along with greater amounts of soil erosion. Large concentrated animal feeding operations (CAFO) in excess of 1,000 animal units increased groundwater issues as more liquid manure is spread on shallow soils over karst limestone. Feedlots adjacent to trout streams allowed large amounts of manure and sediments to enter the water, especially during high flow periods.

All of these issues could be addressed by converting more acreage into managed grazing systems. Producers using this technology reduce sediment and nutrient runoff as well as reduced amounts of herbicide and pesticide issues by replacing row crops with perennial grasses and forbs. Land is divided into "paddocks" restricting cattle access to a small area for a short time with adequate rest periods to allow vegetation to recover. Research has shown that producers using managed grazing systems can show a profit of \$524 per cow versus a profit of just \$132 per cow using a conventional confinement system.

Today the Driftless Area rivals angling opportunities that are found in the Western and some Northeastern streams. Waters that were non-trout in 1980 had naturally reproducing, self-sustaining populations of both Brook Trout and Brown Trout by 2010. Just in the four counties of the La Crosse Area in SW Wisconsin, more than 400 miles of newly classified trout water was added to the "Trout Book" bringing the total to more than 1,000 miles. Numbers in excess of 3,000 trout per mile are not uncommon in streams where only 200 fish per mile could be found just two decades before. A 2017 follow up study of Trout Unlimited's 2008 economic impact found that trout fishing had added another 500 million dollars bringing the total to 1.6 billion dollars. This amount is expected to increase as more local communities realize the positive economic impact of healthy watersheds.

The Demographic and Economic Trends of Trout Anglers: Implications for Recruitment, Retention, and Restoration and Habitat Management Support

Edward Mailett¹, Senior Economist, Richard Aiken², Economist, Matthew Fuller², Economist

¹Division of Economics, U.S. Fish and Wildlife Service

²Wildlife and Sport Fish Restoration Program, U.S. Fish and Wildlife Service

Abstract—Our presentation draws from our 2015 report, Trout Fishing in 2011: A Demographic Description and Economic Analysis (<https://wsfrprograms.fws.gov/>). This report draws from the 2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation to summarize the demographic and economic characteristics of trout anglers. In particular, we focus on how anglers who fished for trout compare to other freshwater anglers who did not fish for trout. We also look at how demographic trends for trout anglers have changed over the previous 20 years. In our presentation we use the findings from our report to consider the potential implications for the future of coldwater fisheries conservation funding.

Background

The National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (Survey) has become one of the most important sources of information on fish- and wildlife-related recreation in the United States. It is a useful tool that not only quantifies the economic impact of wildlife-based recreation but also collects information on participant's socioeconomic characteristics. The U.S. Fish and Wildlife Service (Service) has been sponsoring the national survey every 5 years since 1955 at the request of State fish and wildlife agencies.

The mission of the Service is working with others to conserve, protect, and enhance fish, wildlife, and their habitats for the continuing benefit of the American people. The Service is responsible for national programs of vital importance to our natural resources, including administration of the Wildlife and Sport Fish Restoration Programs. These two programs provide financial assistance to the States for projects to enhance and protect wildlife and fish resources and to ensure their availability to the public for recreational purposes. The Multistate Conservation Grant Program is used to fund the Survey.

Since the 1991 Survey, the Service has produced a summary profile of freshwater trout anglers based on the Survey's findings. Our report seeks not only to summarize the results of the 2011 Survey but to also explore and summarize various trends since 1991 based on a review of all five Surveys conducted during the period 1991 through 2011. We report on the number, effort and demographics of freshwater

trout anglers. We define a freshwater trout angler as an angler who responded affirmatively to fishing in freshwater rivers, lakes, and streams for any species of trout (e.g., Rainbow Trout *Oncorhynchus mykiss*, Brown Trout *Salmo trutta*, Brook Trout *Salvelinus fontinalis*, and Lake Trout *Salvelinus namaycush*). Anglers who reported fishing for these species exclusively in the Great Lakes were excluded to allow results to be comparable to previous studies. Our report also compares the socioeconomic characteristics of trout anglers to other freshwater anglers who did not report fishing for trout. Our data only includes anglers aged 16 years and older due to the sampling protocol of the Survey.

Trout Angler Demographic Characteristics in 2011

In 2011, there were an estimated 7.1 million anglers who reported fishing for trout during the year. In comparison, there were an estimated 27 million freshwater anglers out of a total of 239 million U.S. citizens aged 16 years and older. Thus, approximately 20 million freshwater anglers did not fish for trout during the year. Other species that non-trout anglers may have targeted include many warm water species such as bass, perch, sunfish, catfish, and Walleye *Sander vitreus*. A small number of other coldwater anglers who fished for salmon but not for trout would also be included in this estimate of non-trout anglers.

Of the U.S. population age 16 years and older both trout anglers and non-trout anglers were found to be twice as likely to have fished if they were male

versus female. Specifically, we found that while less than one-half of the U.S. population is male (48 percent), 76 percent of all trout anglers were male as were 73 percent of non-trout anglers. Both trout and non-trout anglers are also predominantly Anglo-Americans. Only four percent of African Americans fished for trout despite making up ten percent of the U.S. population. Asian Americans, who constitute five percent of the U.S. population comprised only one percent of all trout anglers. Also, we found that while 14 percent of the country identifies being Hispanic, only six percent of trout anglers and four percent of non-trout anglers were Hispanic.

When we looked at the distribution of anglers by age, we found that there were proportionally more trout and non-trout anglers between the ages of 25 to 64 years of age compared to the distribution of the U.S. population. Trout anglers tend to skew towards the upper ends of the age distributions. Of anglers 55 years and older, 35 percent of all trout anglers were in this age bracket compared to 28 percent of non-trout anglers. The proportion of non-trout anglers was greater than that of trout anglers for younger ages. Sixty percent of all non-trout anglers are between the ages of 25 to 54 years compared to 55 percent of trout anglers. Younger individuals are less likely to fish for trout or non-trout. Only ten percent of trout anglers and 11 percent of non-trout anglers were under 25

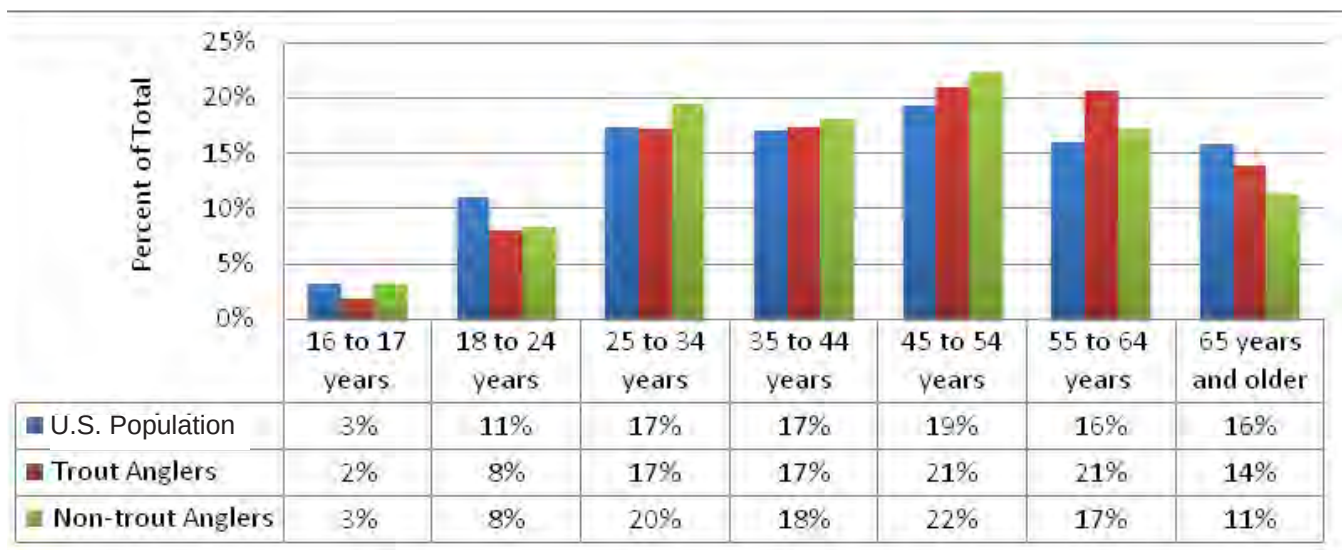
years of age. In comparison, this age group comprised 14 percent of the U.S. population aged 16 years and older. (Exhibit 1)

Trout anglers as a group also tend to have achieved higher levels of education compared to either non-trout anglers or to the U.S. population as a whole. In 2011, 37 percent of all trout anglers had four or more years of college education. In contrast, 28 percent of non-trout anglers had a similar level of education. Overall, 30 percent of the U.S. population aged 16 years and older had four or more years of college education. (Exhibit 2)

Trout anglers are also more likely to have a higher income than non-trout anglers or ordinary U.S. citizens. In 2011, 65 percent of all trout anglers had an income level greater than \$40,000. In comparison, 61 percent of non-trout anglers and 49 percent of the U.S. population had incomes greater than \$40,000. The divergence between trout anglers and non-trout anglers is even greater at higher income levels. For income levels greater than \$100,000, 28 percent of all trout anglers fell into this group compared to 19 percent of non-trout anglers and 17 percent of ordinary U.S. citizens. (Exhibit 3)

In terms of residency, the majority of trout anglers reside in U.S. Census designated urban areas. (<https://www.census.gov/geo/reference/ua/uafaq.html>). The Survey also reported that nearly one-half of all trout

Exhibit 1: 2011 Age Distributions



anglers reside in the western or Pacific states. The number of trout anglers in the mountain states has increased by nearly 20 percent since the previous Survey conducted in 2006. Given that the majority of

trout anglers across the nation come from these states, the percentage increase is quite remarkable compared to other areas that have greater percentage increases but fewer total trout anglers. (Exhibit 4)

Exhibit 2: 2011 Educational Attainment

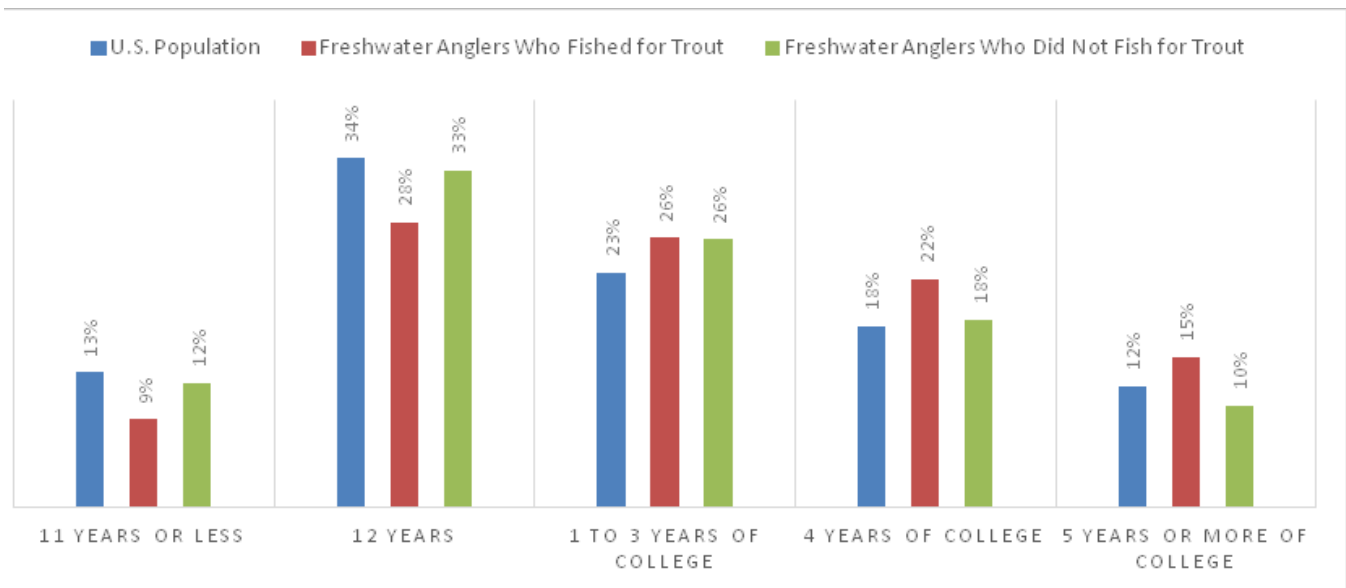


Exhibit 3: 2011 Income Distribution

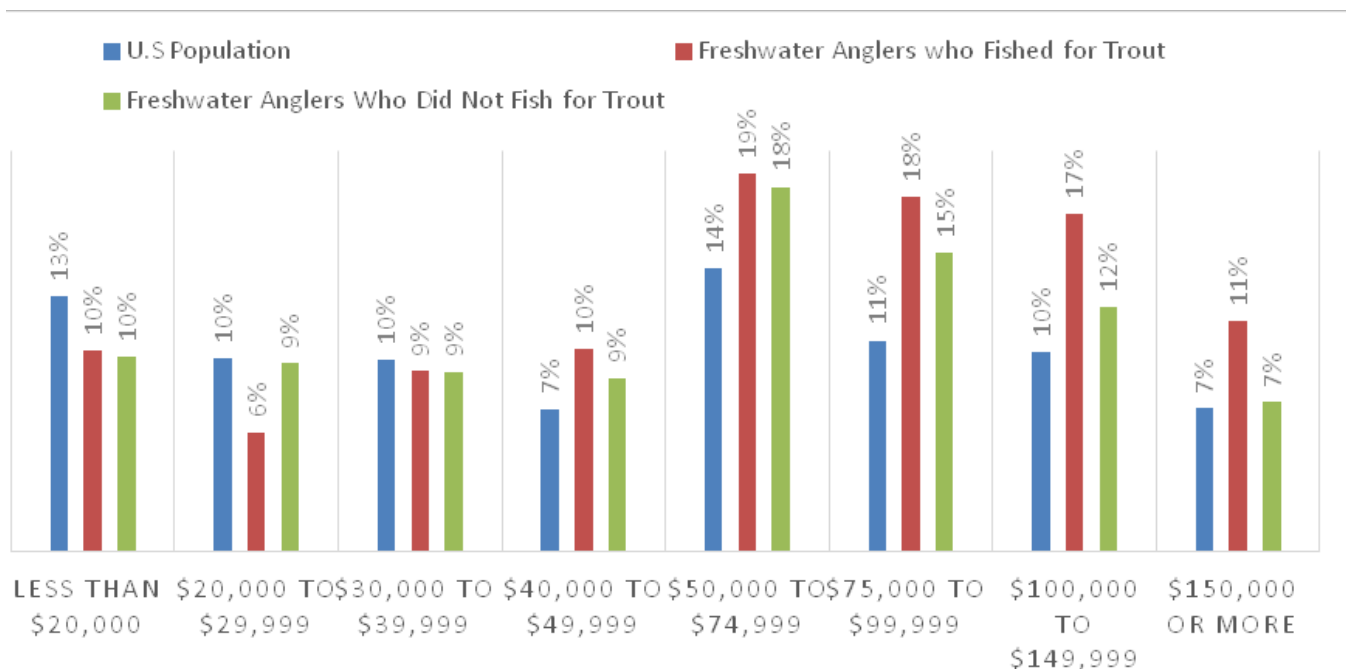
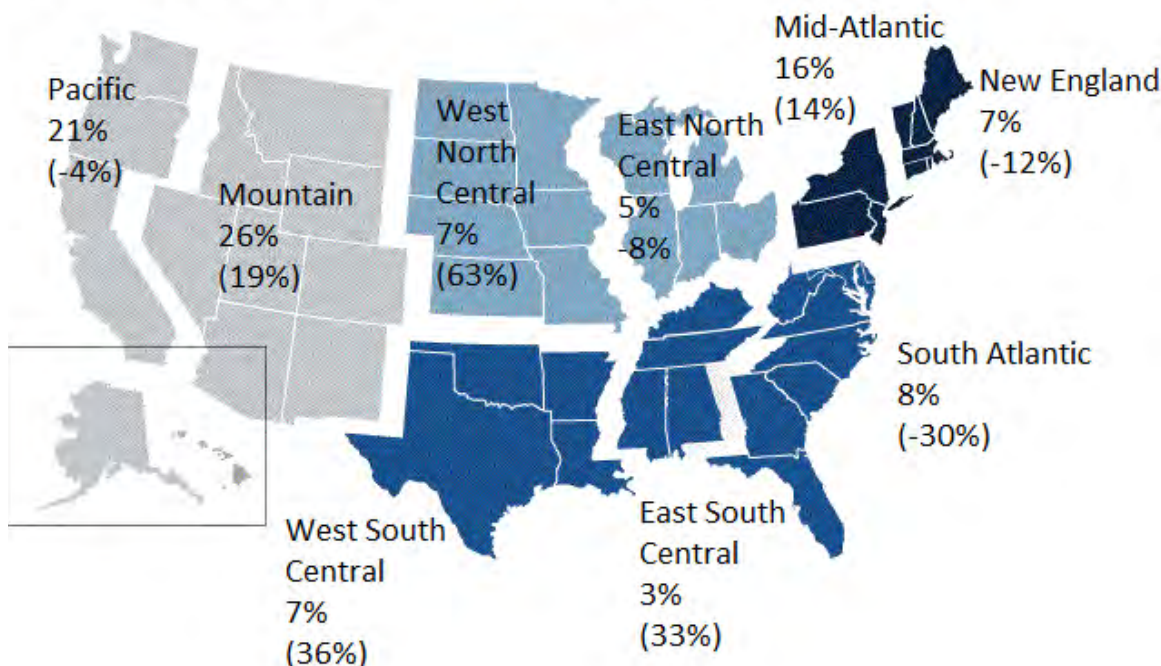


Exhibit 4

Regional Residency of Trout Anglers by Census Division 2011 and Percent Change Since 2006

(Population 16 Years and Older. Excludes Great Lakes Fishing.)

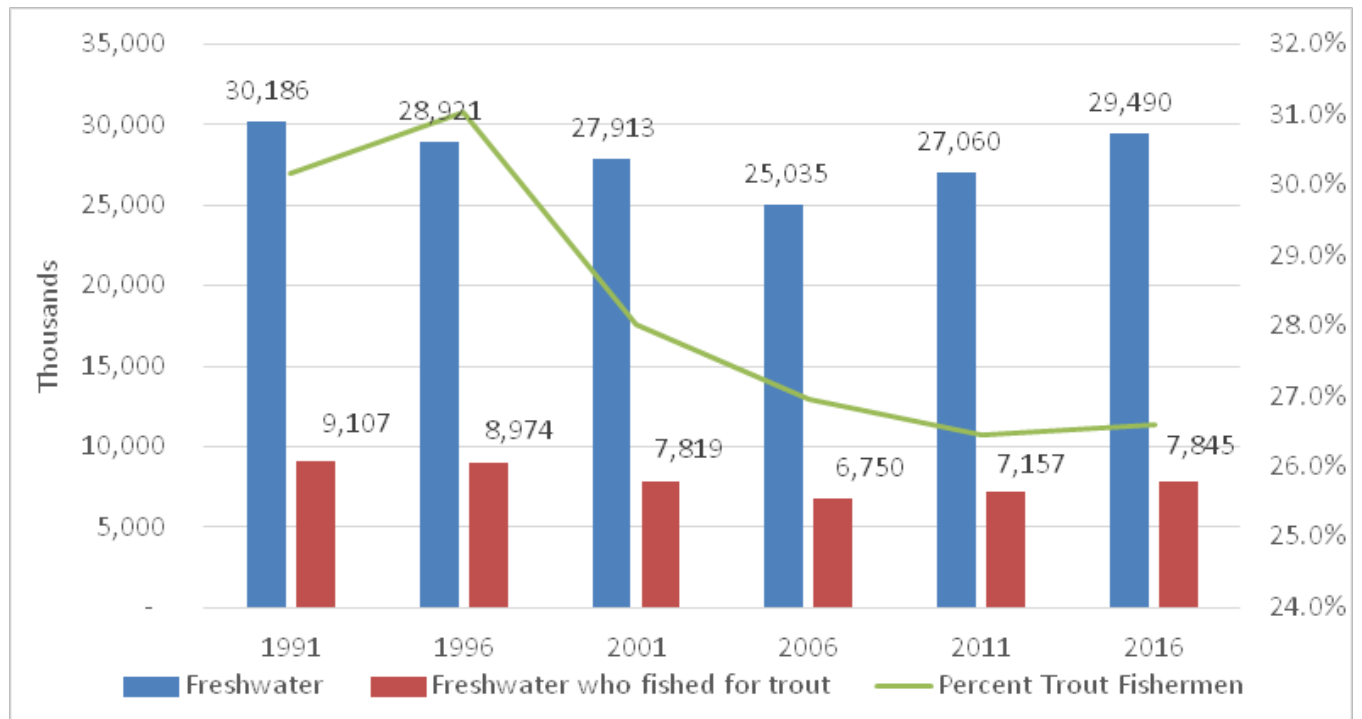
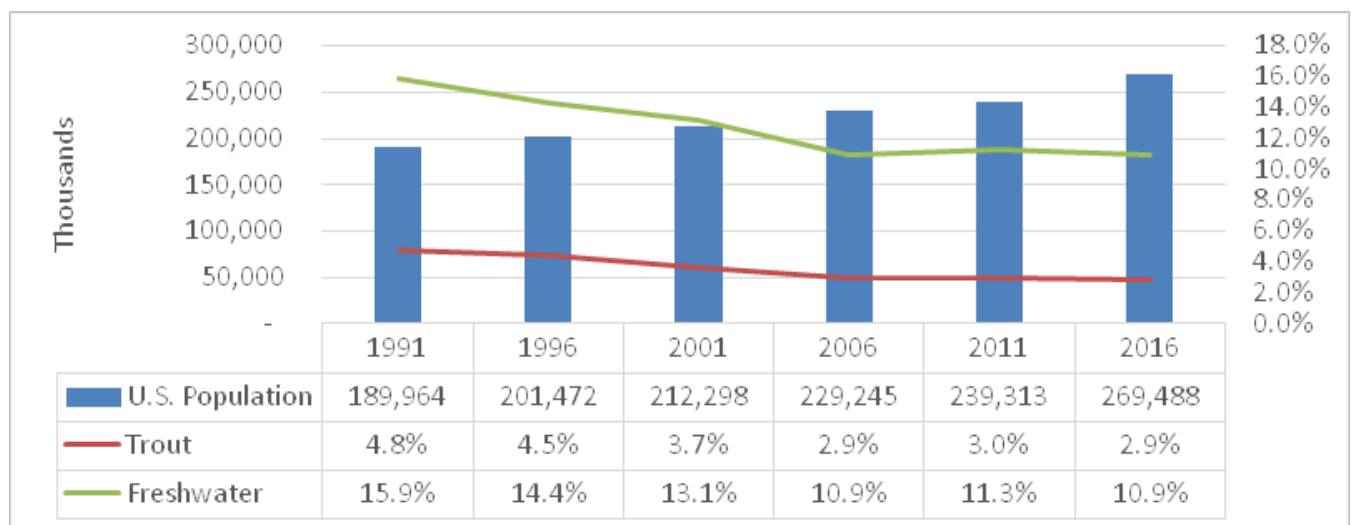


Participation and General Trends

In 2011, 7.1 million freshwater anglers fished for trout during the year. (Preliminary results from the 2016 Survey estimate that the number of trout anglers has risen to 7.8 million.) These two periods (2011 and 2016) represent an increase from the 2006 low estimate of 6.8 million trout anglers. The 1991 Survey reported the highest number of trout anglers, 9.1 million over the six Survey periods. In 1991, trout anglers composed 30 percent of all freshwater anglers. Over time, the percentage of freshwater anglers who fished for trout has gradually declined. In 2016, 26 percent of all freshwater anglers fished for trout. Even though the two most recent Surveys show the number of trout anglers increasing, the percentage of Americans fishing both for trout and freshwater in general has been declining. In 1991, 16 percent of Americans went freshwater fishing and nearly five percent fished for trout. By 2016, 11 percent of Americans fished in freshwaters and only

three percent fished for trout. We note that these trends are consistent with the number of paid license holders reported annually by the States (<https://wsfrprograms.fws.gov/Subpages/LicenseInfo/Fishing.htm>). (Exhibit 5 and Exhibit 6)

Since 1991, trout anglers have increasingly trended to be older in age. In 1991, the Survey estimated that there were 1.6 million trout anglers between the ages of 16 to 24 years. By 2011, the number of anglers has declined to 703 thousand. We see a similar trend for trout anglers between the ages of 25 to 34 years. In 1991 there were 2.6 million trout anglers. By 2011, the number of trout anglers in this group declined by over 50 percent to 1.2 million anglers. In contrast, when we look at the number of older anglers over time we see participation rates increasing. Anglers 55 years and older have steadily risen over the period. In 1991 there were nearly 1.5 million trout anglers in this age group. By 2011 the number of anglers aged 55 and over increased to 2.5 million.

Exhibit 5: National Estimates of Freshwater and Freshwater Trout Anglers**Exhibit 6: Anglers as a Percent of Population**

Correspondingly, while trout anglers have aged over the years, so has their level of educational attainment. In 1991, nearly 50 percent of trout anglers had some college level education. By 2011, the percentage of trout anglers with some college education increased to 63 percent. Specifically, the Survey reported that between 1991 and 2011, trout anglers with four years of college education increased from 1.3 million to 1.6 million, nearly a 20 percent jump. Trout anglers with 5 years or more of college increased from 917 thousand to over one million. In contrast, the number of trout anglers with only a high school education or less declined by over 40 percent, from 4.7 million in 1991 to 2.6 million in 2011.

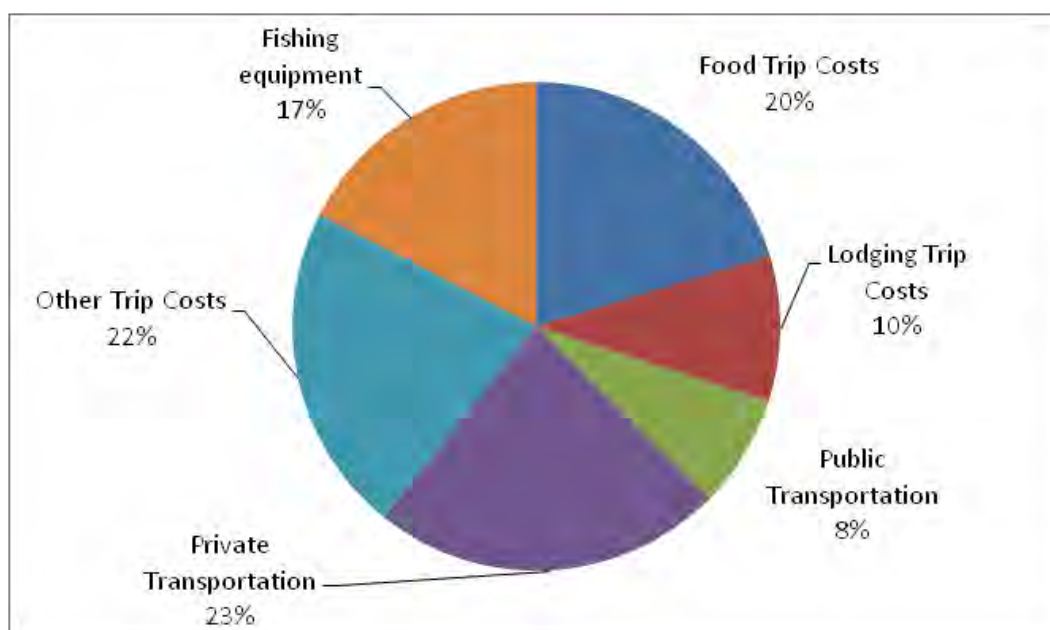
Trout anglers spent a total of \$3.8 billion (\$2016) on fishing equipment purchases and trip-related expenses in 2011. (For the purposes of this study all expenditures were converted into their year 2016 dollar equivalent using the Consumer Price Index in order to account for inflation over time.) Over 80 percent of these expenditures (\$3.1 billion) were trip-related. Between the period 1991 and 2011, total expenditures rose from \$3.1 billion to \$3.8 billion (\$2016) reflecting an increase of over 20 percent. On a per angler basis, the average annual expenditure by trout anglers increased from \$340 in 1991 to \$537 in 2011 (\$2016). The greatest components of trip-related

expenditures were transportation expenses (31 percent of total), food (20 percent of total) and other trip costs (e.g., guide fees, permits and license sales (22 percent of total)). (Exhibit 7)

Trout angler expenditures represent a fraction of total expenditures by all freshwater anglers. In 1991, freshwater anglers spent \$4.3 billion on fishing equipment and \$15.3 billion on trip-related expenses (\$2016) for a total of nearly \$20 billion. This compares to the \$3.1 billion in total expenditures trout anglers spent on trips and equipment. By 2011, trout angler total expenditures on trips and equipment increased to \$3.8 billion compared to a constant expenditure of \$20 billion by all freshwater anglers (\$2016).

Even though there are more freshwater anglers than trout anglers, freshwater anglers still spend more per angler than trout anglers. In 1991, freshwater anglers spent an average of \$634 on trips and equipment compared to \$340 for trout anglers (\$2016). Over time though, average trout angler expenditures have increased greater than for freshwater anglers. In 2011, the average freshwater angler spent \$728 during the year on trips and fishing equipment (a 15 percent increase since 1991) compared to an average expenditure by trout anglers of \$537 (and 58 percent increase since 1991).

Exhibit 7: 2011 Trout Angler Expenditure Breakdown



We also looked at the differences between trout anglers and non-trout anglers in terms of charitable donations. In 2011, trout anglers gave a total of \$22.7 million to freshwater conservation organizations compared to \$114.7 million from non-trout anglers (\$2016). Even though the total number of non-trout anglers was nearly three times as great as trout anglers in 2011, we estimate that the average amount given by a trout angler to a freshwater conservation organization was \$65 compared to twice that amount (\$135) by a non-trout angler (\$2016).

Future Implications

In general, trout anglers tend to be older, more educated, and more affluent than non-trout anglers. Both trout and non-trout anglers also are comparatively homogeneous in race and ethnicity. Could these traits pose concerns for the future of cold water conservation funding support? Freshwater sport anglers generally support freshwater conservation management and programs in one of two ways. Directly via the purchase of State fishing licenses, permits, and stamps or indirectly through an excise tax on fishing equipment purchases and a small motor boats fuels tax.

In 2016, States received \$686 million from the sale of fishing licenses, tags, permits and stamps and \$361 million in grant allocations from the Sport Fish Restoration program (SFR). Since 1991 State license revenues have remained relatively constant in real dollars. In comparison, SFR allocations have been a little more unsteady. In real dollars (\$2016) the States received their highest allocation of \$458.1 million in 2008 and their lowest allocation of \$281.2 million in 1994. Fuels tax likely adds to the greater variability due to the associated variability in the price and demand for gasoline. The real amount of conservation funding has remained flat in contrast to the total U.S. population increasing from 253 million in 1991 to 323 million in 2016 and an economy that has grown even faster, roughly doubling its gross domestic product during the same period.

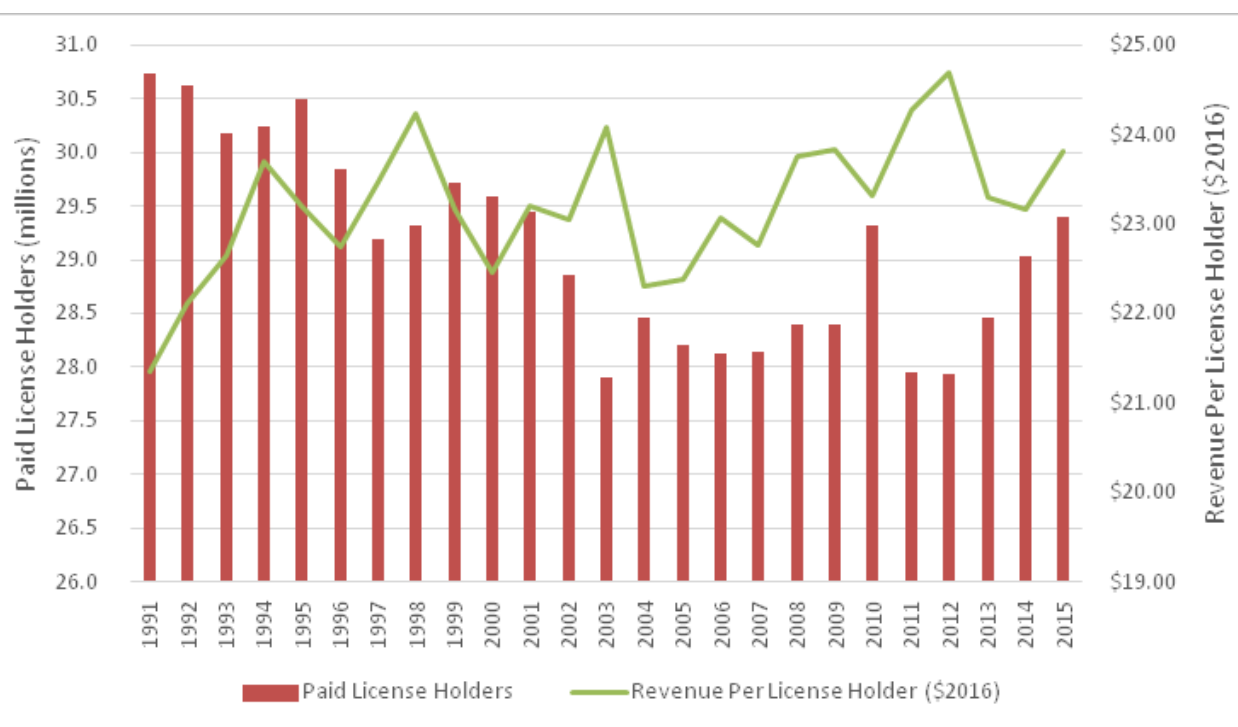
States apparently have maintained a relative constant inflow of license sale revenues via increases in the purchase price of licenses and the imposition of associated permits and fees as the total number of paid license holders has declined. In 1991 there were 30.7 million paid license holders compared to 29.4 million in 2015. The number of paid license holders

has dropped below 28 million several times during this time period, most recently in 2012. In 1991, States on average received \$21.34 per license holder (\$2016), which increased to \$23.80 in 2015. In real terms, obtaining the necessary licenses, tags, permits, and stamps has become more expensive for an average angler over time. (Exhibit 8)

While States have collectively implemented a successful strategy to maintain conservation funding at a stable level, in real terms, during the past 25 years, the demographics and trends of trout and non-trout anglers raises some red flags. Freshwater anglers and trout anglers in particular, tend to be older, wealthier, and higher educated. Of particular concern is how much longer these trout anglers will actively fish and who will fill their shoes (or waders, if you prefer) once they are no longer active. Could the higher real cost of legally fishing be a disincentive in recruiting younger anglers? If so, are our coldwater conservation resources at risk not only from a future reduction in real conservation revenues but also from a future lack of political support out of indifference? Could the greater number of non-trout anglers “outcompete” trout anglers for a larger share of conservation dollars and related projects for warmwater projects if the number of trout anglers declines in real or relative terms?

Another concern is how to effectively communicate the benefits of our coldwater conservation work not just to trout anglers but to the larger number of non-trout anglers and even larger number of non-anglers in order to sustain resources in the future. Industry, which pays the excise taxes for the SFR program, has become increasingly vocal regarding how the funds are being allocated and the “rate of return” associated with chosen projects. Many anglers are unaware that the purchase prices they pay for fishing equipment helps to subsidize industry excise taxes. Our current and collective inability to quickly and accurately summarize the characteristics and accomplishments of projects funded with SFR grants for industry executives and members of Congress raises concerns regarding the efficacy of the program from oversight officials. For example, it is difficult to summarize at a national level the total number of coldwater versus warmwater conservation projects funded, stream miles restored, associated metrics such as enhanced fish populations and associated catch rates, and expansion of recreational fishing opportunities. While the Service is working

Exhibit 8: Paid License Holders and Revenues per Angler: 1991 to 2016



on developing a new data base to collect this type of information, it will only be as useful to answer such questions to the extent that the effort is expanded to summarize and enter such detailed information.

Finally, as the demographics and economic needs of our nation change, it is even more important to be able to understand the characteristics of our sports persons. The Survey is the most important tool for doing so but to be able to understand such metrics as the number of African-American trout anglers by State, the Survey needs to be funded adequately to ensure adequate sample sizes. This funding also comes from SFR grants and has been gradually reduced over the years thus limiting its ability to offer detailed insights into the myriad characteristics it asks of hunters, anglers, and wildlife viewers.

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Imperative Actions for Wild-Trout Conservation Organizations to Combat Climate Change and Increased Water Demand: Unify Anglers, Support Management Agencies, and Drive Water Markets

Rob Van Kirk¹, Brandon Hoffner², Amy Verbeten³, Scott Yates⁴

¹Senior Scientist, Henry's Fork Foundation, P.O. Box 550, Ashton, ID 83420, 208-652-3567, FAX 208-652-3568,

rob@henrysfork.org.

²Executive Director, Henry's Fork Foundation, P.O. Box 550, Ashton, ID 83420, 208-652-3567, bhoffner@henrysfork.org.

³Executive Director, Friends of the Teton River, P.O. Box 768, Driggs, ID 83422, 208-354-3871, amy@tetonwater.org.

⁴Western Water Project Director, Trout Unlimited, 321 E. Main St., Suite 411, Bozeman, MT 59715, 406-522-7291, SYates@tu.org.

Abstract—Wild trout fisheries are threatened by climate change, increased demand for water, and invasive species. However, angler expectations may not keep pace with ecological realities of these changes, resulting in conflicts among angling constituencies, management agencies, and nongovernmental organizations (NGOs) and adding complexity to existing conflicts between wild-trout advocates and water users. We analyzed this complexity in the Snake River basin, which supports both world-renowned wild-trout fisheries and global-scale irrigated agriculture. Climate change and increased irrigation demand have pushed several wild-trout fisheries in the basin to the limit of angler acceptability. Actions that might sustain any of these fisheries into the future often have negative consequences for others. Resulting conflicts pit popular, economically important nonnative-trout fisheries against conservation of native trout and other ecologically valuable resources, potentially placing agencies in no-win situations and fracturing the NGO community. Given political realities, shrinking agency budgets, and grave external threats to wild trout and related resources, conservation NGOs must cooperate with one another to (1) educate, unify, and find pragmatic solutions among all users who value these resources, (2) provide technical and political support to agencies, and (3) develop market-based mechanisms through which wild-trout advocates can exert large-scale influence on water management.

Introduction

Wild trout fisheries throughout the western U.S. are threatened by climate change, increased demand for water, and invasive species, compounding existing effects of dams, reservoirs, and diversion of surface water for irrigation. At the same time, anglers continue to expect the same quality of wild-trout fishing they have enjoyed for many decades, exhorting conservation nongovernmental organizations (NGOs) and fisheries management agencies to take stronger actions to protect their favorite fisheries. When angler desires are not accompanied by broad understanding of larger climatic, socioeconomic, and ecological contexts, demands to restore or maintain traditionally high-quality wild trout fisheries can pit different fisheries against one another and put agencies and NGOs in no-win situations that can result in “least-common-denominator” outcomes. In this paper, we explore the complexities of fisheries management in the upper Snake River basin of Idaho and Wyoming, focusing on new roles for conservation NGOs.

The Upper Snake River Basin

The upper Snake River basin is the 93,000-km² drainage area of the Snake River upstream of King Hill, Idaho (Figure 1). Annual water supply in the upper Snake River basin averages 13,700 Mm³, 46% of which originates upstream of Heise in the Snake River headwaters and 23% of which is supplied by the Henrys (“North”) Fork (Figure 1). The Eastern Snake Plain Aquifer (ESPA), which underlies the Snake River Plain from King Hill upstream to Ashton, is a key hydrogeologic feature of the upper Snake River basin. Surface water—including streams and the agricultural irrigation system—is highly connected with ground water on the ESPA, which is the sole source of drinking water for all cities and residences on the Plain and is a large source of irrigation water for agriculture (Boggs et al. 2010). Water generally flows through the aquifer from northeast to southwest and discharges to the Snake River in the vicinity of American Falls Reservoir and in the Thousand Springs area near Hagerman. Water levels in and discharge

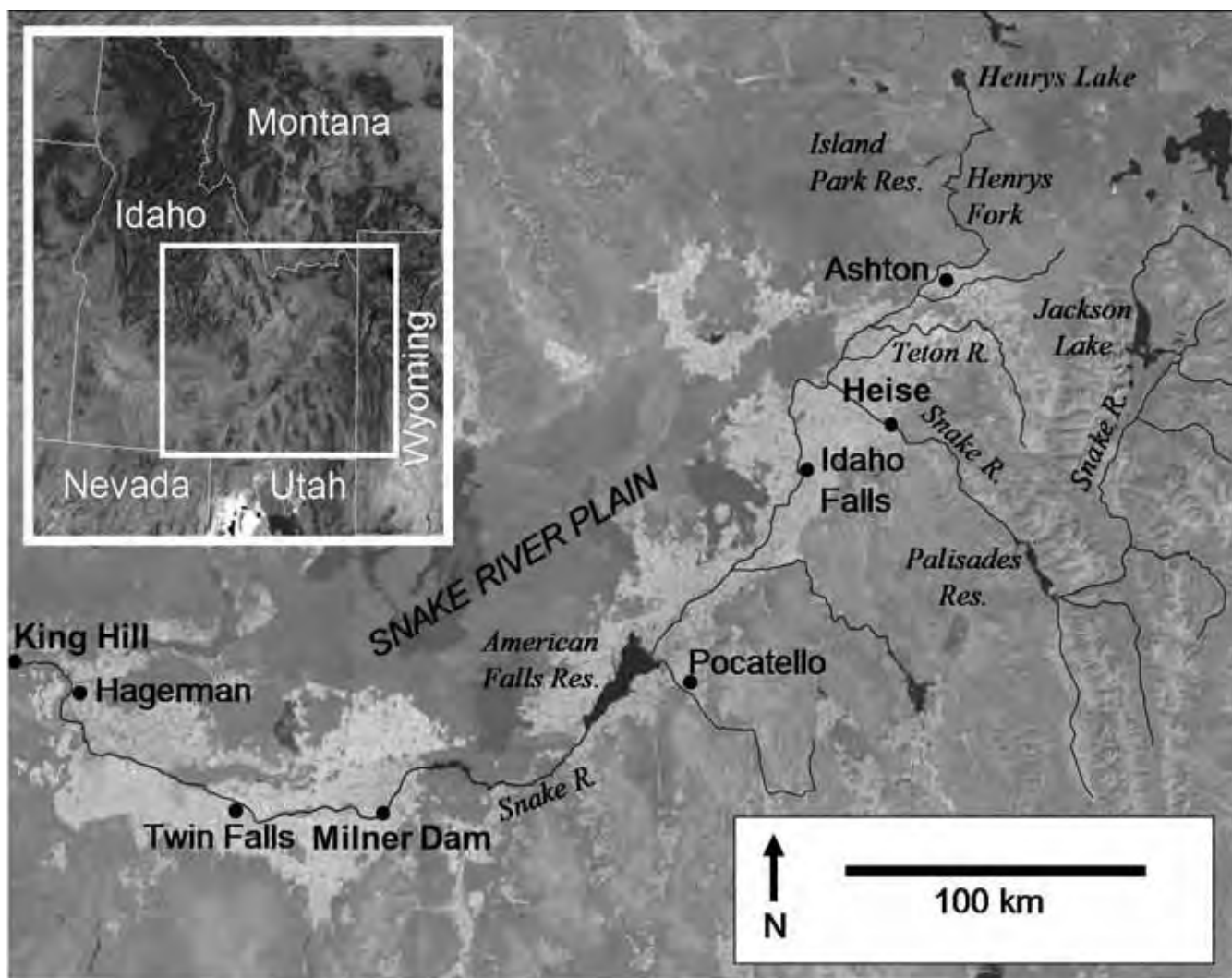


Figure 1. Map of upper Snake River basin.

from the ESPA have been declining for 60 years due to a combination of decreased recharge incidental to surface-water irrigation and increased groundwater pumping (Boggs et al. 2010).

Major Trout Fisheries

The Henrys Fork supports world-renowned wild-trout fisheries for Brown Trout *Salmo trutta* and Rainbow Trout *Oncorhynchus mykiss*, particularly in the 24 RKM immediately downstream of Island Park Reservoir (Figure 1), a reach famous among dry-fly anglers for its prolific hatches and challenging flat-water fishing for large Rainbow Trout (Van Kirk and Gamblin 2000). The Snake River (“South Fork”) between Palisades Reservoir and the Henrys Fork confluence also supports a wild-trout fishery of

international reputation. Although Rainbow Trout and Brown Trout are both abundant in the South Fork, native Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* still support much of the fishery in the upper reaches of the South Fork. Native Cutthroat Trout also provide high-quality angling opportunities for national and international visitors to the Snake River and its headwater tributaries in and adjacent to Yellowstone and Grand Teton national parks in the Wyoming portion of the basin. On the western side of the Teton Range, the Teton River is a destination fishery supported largely by native Cutthroat Trout but also by Rainbow Trout and Brook Trout *Salvelinus fontinalis*. The native Cutthroat Trout populations of the Teton, South Fork, and Snake headwaters are ecologically important and represent a large fraction

of all Yellowstone Cutthroat Trout still present in their native range (YCICG 2016). Other popular national, regional, and local fisheries supported by a combination of hatchery and wild trout include Henrys Lake and other large reservoirs, the Snake River between the Henrys Fork confluence and American Falls Reservoir, and numerous small streams, lakes, and reservoirs. In 2004, nonlocal anglers traveling to the region to fish the Henrys Fork, South Fork, and Snake headwaters contributed US\$46.5 million to the local economy and supported 1,460 jobs (Loomis 2006).

Agriculture

Agriculture is by far the single largest economic sector in the upper Snake River basin, producing an estimated US\$10 billion in goods and services (IWRB 2009). Of 11,750 km² of crop lands in the basin, over 10,000 km² are irrigated. By area, the most widely grown crops are hay (32% of crop land), wheat (23%), barley (16%), potatoes (10%), and feed corn (9%). Most of the hay and corn are consumed by dairy cows in the Twin Falls area, which is now a major international producer of dairy products. On an annual average, around 9,500 Mm³ of surface water and 2,950 Mm³ of groundwater are withdrawn for irrigation. The Thousand Springs area near Hagerman supports a major aquaculture industry. Together, agriculture and aquaculture account for 98% of water withdrawals in the basin.

Water Management and Water-Rights Administration

The upper Snake irrigation system consists of 10 major storage reservoirs, 345 points of surface-water withdrawal, thousands of miles of canals and ditches, and thousands of groundwater wells. Water rights are administered according to a prior appropriation system. The oldest rights are natural-flow rights dating back to the 1870s. Rights to use water stored in the 5,000-Mm³ reservoir system are intermediate in priority, most between 1910 and 1939. Groundwater rights are the most junior in the system, dating from the 1950s through the early 1990s, when a moratorium on additional groundwater irrigation rights was imposed on the ESPA. Administrative irrigation season runs from April 1 through October 31. Once natural flow is insufficient to meet irrigation

demand, storage water is released from reservoirs, usually from late June through mid-October. Refilling the reservoirs during the November 1 – March 31 storage season requires reservoir outflow to be held at very low levels downstream of reservoirs, limiting availability of suitable winter habitat for juvenile trout. Winter flow limits trout recruitment in both the Henrys Fork downstream of Island Park Dam (Mitro et al. 2003) and the South Fork downstream of Palisades Dam (DeVita 2014). Although most reservoir storage occurs during the winter, maximum reservoir storage is reached after capture of snowmelt in the spring and early summer.

Because most of the surface water diverted for irrigation seeps into local and regional aquifers and eventually returns to the river through groundwater pathways, diversions in certain river reaches are maintained primarily by this return flow. Hence, at certain points in the system, the river is either allowed to dry completely or is managed at very minimal flows during irrigation season to limit the use of storage water and allow downstream water-rights to be met primarily with return flows. The lower Teton River is often dry during much of the irrigation season, and only minimal flows are left in the lower Henrys Fork and the Snake River near Blackfoot, reducing trout habitat and water quality during the hot summer months. In fact, the management objective for the entire system is 0 flow at Milner, the downstream-most point of surface diversion (Figure 1). Water rights downstream, including a hydropower right at Swan Falls Dam south of Boise, are met by groundwater returns and tributary inflows.

In 1995, the Idaho Department of Water Resources implemented conjunctive administration of surface and groundwater rights in the upper Snake River basin, which led to water “calls” by senior surface water users against junior groundwater users. After 10 years in the water court system, one of these calls resulted in a settlement between groundwater and surface users in 2015. This settlement requires groundwater users to cut use by 296 Mm³/year or mitigate through managed aquifer recharge or storage rental. In addition to mitigation-specific managed recharge, the State conducts a managed recharge program using surplus natural flow when available, with an annual target of 300 Mm³/year (IWRB 2009).

Stressors on Wild Trout Fisheries

Increased Consumptive Use of Water

Although surface-water diversions have decreased over the past 40 years, increased crop production per unit of water withdrawn, decreased groundwater recharge incidental to irrigation, and increased groundwater pumping have combined to increase total consumptive use of water in the basin over the past three decades, as evidenced by decreased basin outflow relative to inflow (Figure 2). Increased consumptive use and declining aquifer levels have decreased groundwater return flows to the river, reducing streamflows, increasing thermal stress on fish, and requiring delivery of more storage water to meet irrigation demand system-wide throughout the irrigation season. Although decreased groundwater returns have moved hydrologic regimes away from those artificially established by inefficient surface irrigation prior to the 1970s and back toward more natural, “flashy” hydrologic regimes, many wild-trout fisheries in the basin—including those in popular spring creeks such as Silver Creek in the Wood River Valley—have evolved around irrigation-supported groundwater discharge and are now threatened by increases in irrigation efficiency. In particular, sprinkler irrigation increases crop production per unit of water withdrawn but decreases the amount of withdrawn water that is returned to streams.

Climate Change

Although the high elevations and favorable geography of the upper Snake River basin have muted effects of climate change somewhat compared with the Pacific Northwest, California, and Southern Rockies, climate change in the upper Snake River basin is resulting in warmer springtime temperatures, earlier snowmelt, a decreasing fraction of annual precipitation falling as snow, and a longer summer low-flow season (Al Chokhachy et al. 2017). For example, in the high elevations of the Henrys Fork watershed, mean April-June temperature has increased 0.7°C per decade over the past 40 years, moving peak runoff an average of 5.4 d earlier per decade, all other factors being equal (Van Kirk 2017). As an example of direct effects of higher water temperatures, Brown Trout have increased in several reaches of the lower Henrys Fork from less than 10% of the total trout population in the early 2000s to over 30% in recent years (Idaho Department of Fish and Game, unpublished data).

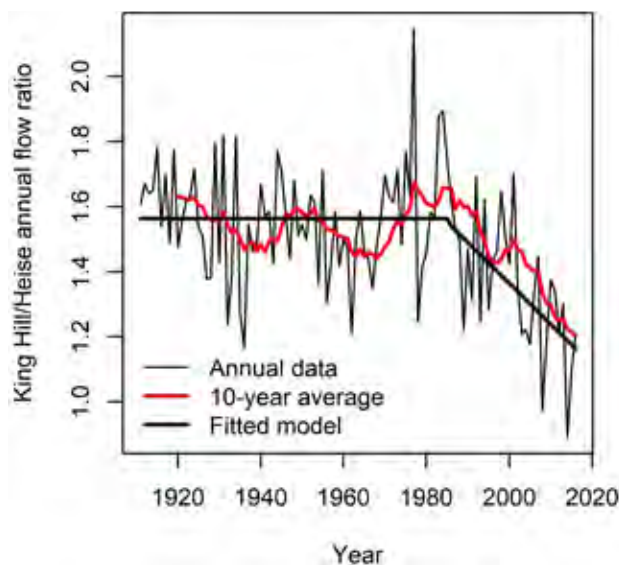


Figure 2. Ratio of annual flow in Snake River at King Hill (bottom of system) to that at Heise (index of total basin water supply).

Continuous water-quality monitoring in those reaches of river has shown that temperatures have exceeded tolerable limits for Rainbow Trout during much of the summer in each of the past 2 years but have remained within tolerable thermal limits of Brown Trout (Henry's Fork Foundation, unpublished data).

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In the upper Snake River basin, where streamflow is highly dependent on management of the reservoir system, the combination of increased consumptive use, early runoff, and decreased snowpack leads to increased reliance on storage water to meet irrigation demand in mid- to late-summer. Increased delivery of reservoir water during the fishing season leads to higher streamflows and turbidity in reaches downstream of reservoirs, decreasing the quality of the angling experience. As mentioned above, increased use of storage results in lower winter flows during refill of the reservoirs after irrigation season, limiting trout recruitment. In addition, increased reservoir drawdown increases suspended sediment mobilization into river reaches downstream and can also result in increased summer outflow temperatures when outflow is sufficiently high to prevent thermal stratification within the reservoir (Figure 3).

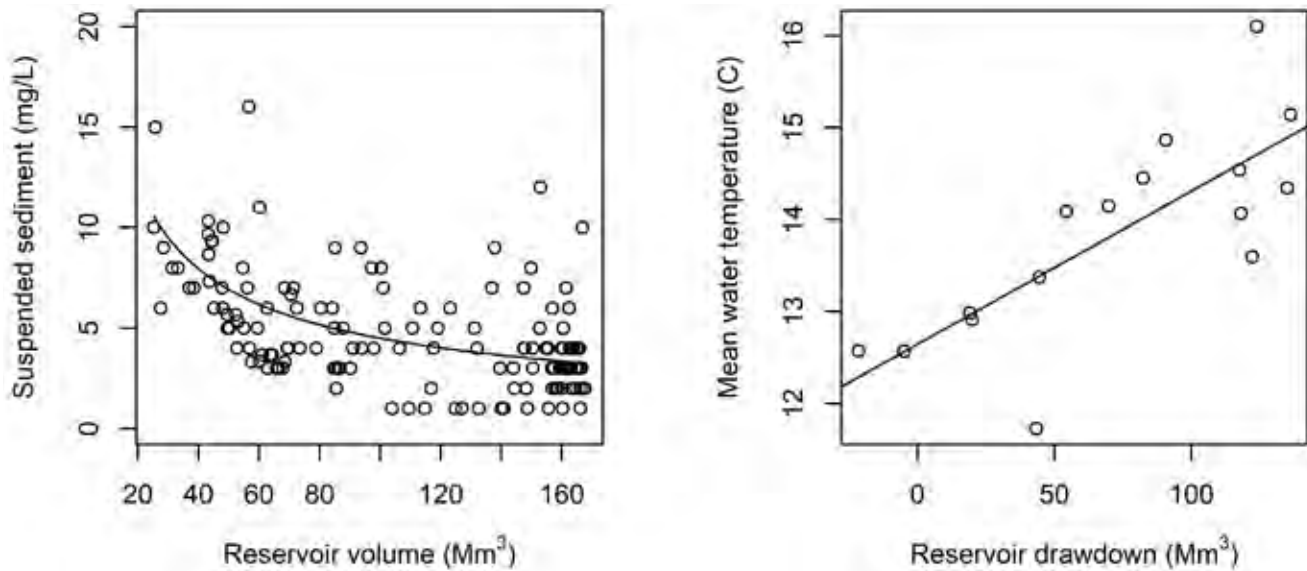


Figure 3. Suspended sediment concentration in Henrys Fork as a function of Island Park Reservoir volume (left panel), and mean May 1-August 31 water temperature in Henrys Fork as a function of irrigation-season drawdown of Island Park Reservoir (right panel).

Invasive Species

Invasions of nuisance mussels threaten fisheries in the upper Snake River basin, especially given the presence of these species immediately to the south in Utah and to the north in Montana. Boat inspections and education have prevented spread of these mussels into the upper Snake River basin thus far, but continued vigilance will be required to prevent spread of mussels and nuisance aquatic plants into quality trout waters in the upper Snake River basin. Invasion of undesirable fish species into waters that traditionally have supported wild-trout fisheries may be more difficult to prevent, given that warmwater species are already present in many lakes and reservoirs in the basin. Nonnative Rainbow Trout and Brook Trout already threaten remaining Cutthroat Trout populations in the basin (Meyer et al. 2014), and some anglers even consider displacement of Rainbow Trout by Brown Trout in low-elevation river reaches as an undesirable invasion. As the climate continues to warm, such invasions are inevitable, potentially leading to a succession of fish species similar to that in the lower Henrys Fork over the past four decades: Cutthroat trout, Rainbow Trout, and now Brown Trout. Only time will tell if warmwater species will eventually displace Brown Trout in these low-elevation river reaches.

Management Challenges

As water supply becomes limited, water quality deteriorates, wild-trout recruitment and survival decrease, and warmer temperatures favor invasion of Brown Trout and potentially warmwater species, anglers are becoming increasingly concerned about the future of their favorite wild-trout fisheries, particularly those for Rainbow Trout. Anglers who are passionate about a specific type of fishing on a specific water body have been especially vocal. In most cases, improving a specific fishery would require water-management actions to improve stream flows and/or reservoir levels. However, actions taken to improve one fishery may harm another. Several such examples have come to light in recent years. For example, a large drawdown of Island Park Reservoir can be somewhat offset by increased delivery out of Henrys Lake upstream, pitting water quality and quantity in Henrys Lake against that in and downstream of Island Park Reservoir. Another example is the challenge of increasing summer stream flows in the lower Teton River and lower Henrys Fork to benefit fisheries there. Doing that requires increased delivery of water from Island Park Dam, to the detriment of the popular fishery in the river reach downstream of the dam.

A third example highlights conflict between native fish conservation and popular nonnative trout fisheries.

During dry years, some of the storage water released from Island Park Dam is delivered to the Teton River to support irrigation diversion there. Construction of Teton Dam in the mid-1970s was intended to provide storage in the Teton watershed, which would have negated the need to deliver Island Park storage to the Teton River. However, that dam failed, and there is currently no storage reservoir on the Teton River. Thus, the need for storage delivery to the Teton River remains. A recent U.S. Bureau Reclamation planning process identified some storage options to partially replace storage lost when Teton Dam failed (USBR 2015). This storage would reduce or eliminate the need for delivery of Island Park storage water and thus benefit popular nonnative wild-trout fisheries on the main-stem Henrys Fork. However, all of those storage options would have had at least some negative effects on native trout in the Teton River, and none proved to be acceptable to all stakeholders. There are numerous other examples of water-management options that would improve one fishery in the basin at the expense of another.

Actions for Conservation Organizations

In an era of decreasing agency budgets at both State and Federal levels, restricted ability for scientists and managers in some agencies to openly address issues related to climate change, and increasing complexity of issues facing wild-trout fisheries, conservation NGOs are finding themselves with new challenges but also new opportunities. Our particular three NGOs have responded by collaborating with each other and with other NGOs much more closely over the past three years than we had previously. We have held numerous strategy sessions to discuss the challenges listed above—sometimes with our agency partners and other times without. In addition to a general strategy to share resources and divide workload among ourselves to maximize our collective effectiveness in advocating for wild trout and the water they need, we have identified several imperative actions that are driving much of our current work.

Educate and Unify Anglers

Education and outreach have always been a component of the work of conservation NGOs, but these roles are becoming increasingly important

as the quality and fate of wild-trout fisheries in the western U.S. depend more and more on complex interactions among climate, water quality, water management, water law, water-rights administration, and regulatory and legal processes. For anglers to be effective advocates for public resources, they need to have accurate information and a good understanding of the biological, physical, socioeconomic, legal, and political factors that affect their favorite fisheries. Moreover, they need to understand the larger context in which their fishery exists and how actions to “fix” a problem in one fishery or water body often have negative consequences for another fishery that has an equally passionate constituency. Identifying values, threats and solutions that are common to all anglers has proven to unify rather than fragment anglers. Recent discussions about transfer of federal lands in the West to the States, for example, has unified traditionally disparate groups of outdoor enthusiasts behind the common cause of retaining public access for fishing, hunting and other recreation.

Although it is challenging to convey complex concepts to people who mostly “just want to go fishing,” we have found that using multiple electronic media and communications strategies has been more effective than the traditional approach of a quarterly newsletter and a few public presentations to engaged members. Most NGOs now have professional education staff in addition to more traditional “communications” staff and participate in educational programs such as Trout in the Classroom for school-age kids and the Idaho Master Naturalist program for older adults. Most of these educational programs are conducted in cooperation with State fish and game and other agencies.

Support Government Agencies

Whereas more traditional models of interaction between NGOs and agencies involved NGO advocacy as part of agency public-review processes, many conservation NGOs are now enhancing the effectiveness of resource-management agencies through technical, financial, and political support. Many conservation NGOs now have in-house science and monitoring programs, which can contribute valuable site-specific scientific information beyond what agencies can collect with limited resources and budgets. Especially when site-specific studies are designed and implemented in cooperation with

agencies, the results can be cited by agencies to inform management decisions. All three of our NGOs routinely provide technical and scientific support to State and Federal agencies in the areas of native trout status and distribution, water-quality monitoring, and hydrologic modeling. In many collaborative research and restoration projects, agencies can contribute equipment and staff expertise to on-the-ground projects implemented by NGOs, maximizing conservation results per unit of resource expenditure. In other cases, NGOs can apply for and receive grant money that can be used to implement projects on behalf of agencies, again increasing the effectiveness of both entities. Finally, NGOs can provide political support to agencies both directly through public-review processes and indirectly in situations where agencies do not have mandates to serve as true advocates for the resource. An example of the former is when NGOs participate in public-review processes to demonstrate support for local or regional agency actions when that support is necessary for the action to receive approval at higher levels within the agency. A recent example of the latter is involvement of our NGOs in negotiations over new water-right applications. Although our agency partners provide technical input to the negotiation process, NGOs have greater latitude to more strongly advocate for the resource at the negotiating table.

Influence Water Management Via Markets

After carefully analyzing the limited options available for moving existing water around the basin differently to benefit fisheries resources, we have concluded that the zero-sum approach to water management will, at best, benefit one fishery at the expense of another. Hence, our three organizations have adopted a strategy of decreasing consumptive use of water through market-based mechanisms. The recent settlement between groundwater and surface water users has opened up a new market in which groundwater users are willing to rent water for groundwater recharge, to be applied toward mitigation. In this new market place, we seek to facilitate exchanges that reduce irrigation diversion in key stream reaches, both to keep more water in these reaches during the summer and to limit the need for delivery of reservoir storage. The exchanged water would be diverted for managed aquifer recharge in the fall, winter or spring, when physical delivery of storage water would not be needed.

While NGOs cannot replace government agencies with legal authority to manage fisheries, NGOs must increase their roles in education, technical services, and water marketing to ensure the future of wild trout.

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Session 1

Anglers, Stakeholders and the Socioeconomics of Wild Trout





2017 Study Measures Economic Impact of Trout Angling in the Driftless Area: How will it Inform Wild Trout and Other Resource Management Decisions in the Region?

John (Duke) Welter, Outreach Coordinator

Trout Unlimited Driftless Area Restoration Effort (DARE)

Abstract—A new economic impact study of trout angling in the unglaciated region of the Upper Mississippi River Basin reports anglers spend close to \$1 billion per year in the region and spinoff economic benefits bring the total over \$1 billion. The report bolsters the conclusions of a 2008 study that reported a \$1.1 billion impact. In addition, the new study estimated over 6,500 jobs across the region are supported by angling visitors. Demographic information suggests that more women are visiting the region to fish trout streams, that the angling population is aging but is willing to take numerous multi-day trips per season and travel significant distances to enjoy quality trout angling. Anglers reported they favor fishing for wild Brown Trout *Salmo trutta* and Brook Trout *Salvelinus fontinalis* than stocked Rainbow Trout *Oncorhynchus mykiss*, and that the primary reasons they prefer the region are (1) high-quality trout fisheries resulting from growing restoration efforts and (2) abundant public access and generally hospitable landowners across the region.

Introduction

A recently-released 2017 economic impact study reaffirms the extent of angler spending for trout fishing in the Driftless Area, building on an earlier study from 2008. Its conclusions could help build a trove of helpful information on angler preferences and economic impacts, which in turn can lead to increased support for wild trout management.

The Driftless Area of the Upper Mississippi River Basin encompasses 24,000 mi² of western and southwestern Wisconsin, southeast Minnesota, northeast Iowa and northwest Illinois. It is so named because past glaciers bypassed the area and did not leave behind the boulders and gravel known as “glacial drift”. The Driftless Area is underlain by limestone and sandstone bedrock making up scenic bluffs and ridges as high as 500 ft. Through its valleys wind over 6,000 mi of spring-fed creeks and rivers which can provide excellent fishing for trout if adequate habitat is provided.

This area’s lands and waters suffered from hard and uninformed land use after European settlement in the 19th century. Tremendous erosion from the bluffs and hillsides deposited thick blankets of sediment on valley floors. Many communities were deluged by repeated flooding and sediment. Some moved to higher ground and others were simply buried and abandoned. By the 1930s, many communities were using their entire budgets to replace bridges washed

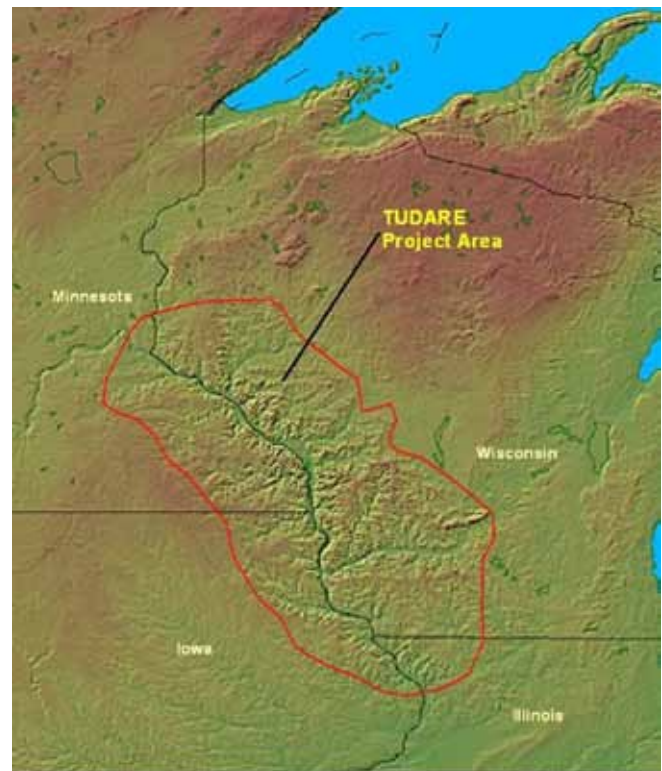


Fig. 1: Map of Driftless Area (in red boundary) with landforms and state boundaries.

out by numerous floods and clear out mud-clogged roads. Upland erosion control efforts began in the 1930s and have stanchd much of the soil loss off the uplands, but extensive streambank erosion has continued in many of the area’s 600 watersheds.

(Leopold 1935, Trimble 2012) Despite upland soil conservation efforts beginning in the 1930s, by 1958 the Fisheries Bureau Director for the Wisconsin Conservation Department (now Department of Natural Resources, or WI DNR) studied the area and concluded that poor land use continued to so challenge streams that in the future it was unlikely that trout populations would be able to persist (Brasch 1958, from Personal Communication, Vetrano, D. WI DNR)

Stream restoration efforts began in the 1970s and have continued, but a more widespread program, the Driftless Area Restoration Effort, began in 2004 and has greatly expanded those efforts. Since 2004, public fishing access has increased from 750 to over 1,200 mi, and many landowners do not grant easements but are amenable to allowing access to anglers who request permission. As a matter of policy in both Wisconsin and Minnesota, Departments of Natural Resources and Trout Unlimited require legal public fishing access on any stream where those entities' funding is used for restoration. As a result of those policies and ensuing restoration, many more high-quality trout fisheries are available to anglers, and more visitors explore the region each season.

How do those angling visitors spend their money when they visit, and how does that spending impact the economies of the rural communities across the Driftless Area? A 2017 study sought answers to those questions, and gave some basis for comparison to an earlier study done in 2008. We will examine the new study and the lessons it offers for local businesses, tourism groups and the economic development community. In turn, it is hoped that knowledge can help inform policy makers and natural resource managers in their approach to supporting management of trout resources in the region.

Historic Impacts of Driftless Area Watershed Restoration

Spring creeks in the Driftless Area have naturally-occurring features which make them attractive habitats for trout and other coldwater species. They flow year-round from springs and seeps at the base of the sandstone-limestone ridges of the region. Their temperatures consistently hover at around 48 °F and their flows are not significantly affected by drought periods. The emitted water has fallen as rain or snow on the ridgetops and hillsides and been absorbed

into the aquifer to filter through the limestone and sandstone bedrock to emerge from the springs. The bedrock leaches calcites which contribute to a high pH, which in turn contributes to fertile water chemistry for growth of aquatic animal fish and plant life.

However, decades of deleterious land use contributed so much eroded soil to these systems that most streams flow through incised channels edged with eroding banks. That bedload chokes off instream habitat such as riffles, fills deeper pools, and reduces the nutrients available to the biotic community. In addition, farmers have avoided plowing near streams to keep horses and later tractors from collapsing banks. That led to development of wooded streamside corridors of box elder, soft maple and ash. Those corridors prevent sunlight from reaching the stream and discourage growth of aquatic vegetation.

Although the hard-used streams look like ribbons of blue water and greenery, they are actually austere biological systems. Without riffles and instream vegetation, insect growth is limited to a few species and a few individuals, all tolerant of degraded conditions. While trees may fall into the creek when their shallow roots are undercut by flood waters, their falling collapses banks and contributes more sediment to an already sediment-choked system. Pools fill with sediment and raw eroding banks provide no overhead cover, essential habitat for trout.

Current restoration practices used across the Driftless Area work to reverse these adverse impacts and take advantage of the intrinsic positive aspects of these systems. Fisheries managers and river scientists working in other parts of North America may find some of these practices, such as tree removal, to be contrary to their assumptions about “what trout need”. However, in these badly-degraded systems with cold spring flows, many of those assumptions don't seem to apply. Removing the shallow-rooted tree species (while working to retain oaks, willows and cottonwoods) opens up the stream corridor and encourages growth of aquatic vegetation as well as grasses and forbs on the corridor. Instream vegetation provides abundant food species and riffles provide even more (Personal Communication, D. Vetrano, WI DNR). Bank sloping recreates a flood plain which, when revegetated with deep-rooted native grasses and or other soil-holding turf, absorbs flood flows and reduces bank erosion. A healthy flood plain can absorb flood waters as well. Nearshore vegetation

dramatically expands inputs of terrestrial food species to the stream, not just the “hoppers-beetles-cricket-and-ants” recommended to fly-fishers, but also bees, wasps, flies, and other insects which can become trout food. Various kinds of instream structures have been used to provide worthwhile habitats. Rootwads in the stream provide habitat for insects, small and large fish species. Vortex weirs encourage pools for hot weather and winter refugia. Constructed underbank structures can provide overhead cover, but are gradually being replaced by other means of creating that cover such as wood or deep-rooted prairie grasses. Side channels and backwaters provide habitat for frogs, turtles, minnows and other species.

With these restoration practices, managers observe scouring of sediment, dramatic increases in invertebrate populations and attendant increases in populations of trout and other fishes. A degraded stream such as Gilbert Creek in Dunn County, Wisconsin, before restoration activities began had only 100 trout/mi in pre-restoration surveys, while after the restoration nearly 2,500 trout/mi were found. It is only one of several score examples to be found across the region (Personal Communications, M. Engel, and J. Sours, WI DNR).

Restored streams are also much more able to offer optimum conditions for trout reproduction. Exposed gravels offer redd sites and the diversity of habitats encourage small and larger trout populations to thrive. In many restored waters in the Driftless Area, no stocking has been done for years; in others, some supplemental stocking using fingerlings from feral brood stock is done on a limited basis, mostly in streams where reproduction is still limited for some reason.

During the past 10 years approximately \$5 million a year has been invested in restoration in Wisconsin and Minnesota Driftless Area streams. Trout Unlimited’s Driftless Area Restoration Effort (TUDARE) has, since 2004, worked to expand funding and capacity for restoration work. With its partners, DARE gathers funding from a variety of sources: Trout Stamp dollars, Farm Bill conservation grants from U.S. Department of Agriculture’s Natural Resources Conservation Service, U.S. Fish & Wildlife Service for endangered species habitat, Minnesota’s Outdoor Heritage Fund (sales tax) dollars, foundation grants, private contributions, donations from Trout Unlimited groups and local conservation groups, and other sources (Personal Communication, J. Hastings, TUDARE Project Manager).

Studying Angler Preferences and Recreational Impacts of Restoration

Healthy trout populations draw anglers, and expanding the number of quality opportunities increases the number of visitors. Just how much of an economic impact they engender has been an ongoing area of study in the Driftless Area for over 25 years. Anecdotally, each year since TUDARE began we have observed more and more out-of-state cars parked at access sites across the Driftless Area. The 2017 and 2008 surveys have confirmed the expanded area from which Driftless anglers travel, and what they spend while they are in the region. While some methodological differences limit the comparability of the 2008 and 2017 surveys, let us examine and compare those findings as much as possible. We will list the respective findings with the 2008 survey finding first, and the 2017 finding second.

Angler preferences were examined in both surveys. Who are the anglers traveling to the Driftless Area to fish? Are those attributes changing from the 2008 to the 2017 survey? From where do anglers travel to visit and fish in the area? How far are they willing to travel? How long do they stay, and where do they prefer to stay? What do they spend while in the area? How often do they visit? What factors contribute to the overall economic impact of these recreational trout anglers? What are the most important attributes that draw them to the Driftless Area?

Augmenting the survey findings, this author interviewed a number of anglers and businesses operators earlier this year to develop a less statistical, more personal picture of the impact of visiting anglers. An article published in the quarterly magazine, *Wisconsin Trout*, January 2017, summarized those interviews and will be drawn on here.

A few notes on the methodology for the 2017 study are in order. From lists of holders of trout stamps (required of all trout anglers) in Wisconsin, Minnesota, and Iowa, a random list of 1,500 anglers was developed. A mailed survey was sent to the entire list. Some anglers responded to the survey online, even if they were not included on the list. From those contacts, 301 surveys were returned. The angler population and survey response rates were comparable to the 2008 survey. Note that responses from anglers residing inside the Driftless Area itself were not used in the calculation of the economic impact by the

consultant. However, for the purpose of comparing overall angler economic impact to the 2008 study their information was evaluated

The survey instrument was developed by the consultant, Prof. Donna Anderson, Ph.D., Professor of Economics at University of Wisconsin-La Crosse, and results were analyzed by her. In addition to addressing questions included in the 2008 survey, she was also asked to evaluate the number of jobs across the Driftless Area supported by recreational angling and to take a look at the “trout economy” of a number of area communities which have developed a focus on angling and recognize its value to their economies.

What do visiting anglers identify as the chief “draws” to fishing in the Driftless Area? Major draws for visiting anglers are wild trout (66%), abundant public access (55%), high-quality trout streams (49%), and restored waters (46.6%). Over 88% of anglers are aware of restoration efforts and over 80% say restored waters will keep drawing them to the area.

Two aspects, “Friendly landowners” and “Opportunities to Catch Stocked Trout” were worth exploring. About 1,200 of the 6,000 mi of trout streams in the region are covered by public fishing easements and that number increases each year. That leaves a significant number of streams where permission is necessary to fish a stream (e.g., Iowa) or where permissive access is helpful but not legally necessary (e.g., Wisconsin and Minnesota). Courteous anglers find hospitable landowners in most cases, and can expand their fishing opportunities. However, with the de facto requirement that any restoration take place on streams with an access easement, the number of miles of accessible water is expected to continue to increase.

Catching stocked trout seems to be more important to anglers in one state, Iowa, than in the other two where wild trout are most favored. Iowa stocked some 260,000 catchable sized Rainbow Trout *Oncorhynchus mykiss* in its streams in 2012 and that number has likely increased. While anglers in other states, when asked their preferred trout to target, most often prefer wild Brook Trout *Salvelinus fontinalis* or Brown Trout *Salmo trutta*, (e.g., in Minnesota’s Root River, a 2006 creel census found 39 % of anglers preferred Brown Trout to 7% who favored Rainbow Trout), a component of the population still expressed a preference (39%) to catch those Rainbow Trout, predominantly found in Iowa Driftless streams.

Preferred angling method for 51.4% of anglers was fly-fishing, while 34% were spin fishers. Live bait was preferred by 24%. Since many anglers fish with more than one method, totals exceeded 100%.

What are the demographic characteristics of the typical Driftless Angler? They are likely to be male (89%), married (71%), median age 51.3 years, overwhelmingly white, with a median household income of \$80,000 - \$100,000. Over half of respondents have earned a bachelors or higher college degree. Median age had risen from 47 in 2008, comparable to other studies of the angling and hunting population in the United States.

Women anglers appear to be increasing in the area, if figures from the 2008 (7%) and 2017 (11%) surveys accurately measure their involvement. An earlier and more limited measure can be found in a survey of anglers in the Kickapoo River (one of the major rivers in the Driftless Area) from 2001 is an indicator. That survey found women made up 5% of anglers in that watershed (Anderson 2001).

From where do anglers come to the Driftless Area? The average angler traveled 138 mi, or about 2.5 h, one way to fish in the area. That comports with past angler surveys and anecdotal information indicating significant numbers of traveling anglers come from Milwaukee, Chicago, Minneapolis/St. Paul or Des Moines, urban areas within 1-4 h of the Driftless Area. The average angler made 5.84 trips to the region, and brought 2.23 other anglers (median age 42 years) along for 2.44 d. In order, the four most popular months for visits were May, June, September, and April.

What amenities do anglers prefer when they visit the Driftless Area? Lodging choices included camping (34.7%), motels (22.8%) or rented cabins (15.4%). These choices changed somewhat since the 2008 study, with a significant increase in use of rented cabins (8%) and slight decreases in motels (30%) and camping (38%). While many small campgrounds exist across the region, each year it appears more rental cabins are being built and offered to anglers on websites catering to anglers. One recent advertisement for a rental house near an Iowa trout stream asked rates of \$450/weekend night. (Airbnb.com/ /rooms/19820987?; Dorchester, Iowa.)

Economic impacts were conservatively estimated, according to Dr. Anderson. Methodology played a role, as a significant portion of Wisconsin’s trout stamp holders were excluded. Also excluded were

trout stamp holders from Illinois, from which a significant portion of Driftless anglers are attracted. Anglers whose addresses were inside the Driftless Area were excluded, even though those anglers (1) needed to make a decision whether to fish in Driftless Area streams or in other waters, and (2) contribute significantly to the spending and economic impact of angling in the region. For purposes of comparison to the 2008 study, Dr. Anderson made a calculation of economic impact of both intra- and extra-Driftless Area anglers. The upshot of her comparison showed that in contrast to the 2008 survey's conclusion that total economic impact was \$1.1 million, by 2017 it has risen to \$1.6 billion. Because of the differing methodologies, however, the newer figure leads us to cautiously assess the total impact as somewhere between \$1.1 and \$1.6 billion.

The 2017 report concluded that 6,597 Driftless Area jobs were supported by recreational trout angling, primarily in food service, lodging, grocery and liquor stores, gas stations, and other retail.

Dr. Anderson also included an estimate that just over \$5 million is invested each year in stream restoration by a variety of agencies and nonprofits across the region. Those dollars also have their economic impact because those dollars are spent in communities for materials, fuel, equipment, and other purchases. A billion dollar annual return on a \$5 million investment would make a Wall Street profiteer blush.

Environmental benefits and community benefits of restoration extend far beyond the assessment of impact based on spending profiles. These benefits include reduced erosion, reduced flooding impacts, improved water quality, improved fisheries, and a reduction of nutrients often carried into streams and rivers by widespread runoff. They touch communities when water is cleaner and does not have to be treated as intensively for public drinking water supplies, or when infrastructure damages from flooding can be reduced. Further research to assess these benefits should be considered in the future.

How can this economic impact affect trout management in the Driftless Area? As a generator of economic development, recreational fishing does not require much public investment in infrastructure and can help diversify the economies of small rural communities now primarily dependent on agriculture.

A town that recognizes the benefits of bringing in outside dollars through the “trout economy” can work to encourage development of assistive resources. For instance, communities could develop web-based sites with maps of fishing locations, lists of lodging operators and cabins, and businesses that cater to angling visitors.

When issues arise that could adversely impact coldwater resources, e.g., proposed industries that would discharge pollutants into surface or ground waters, it is helpful to remind decision-makers that those waters are essential to the local economy. Businesses that do not directly focus on fishing but benefit from the visitors who come to enjoy that activity can become allies in the efforts of angling groups to protect and restore these waters. Local voices carry more weight in the development of protective and restorative measures than do voices from farther off.

When people familiar with the benefits of restoration in the Driftless Area—not necessarily anglers—speak up for continued restoration through programs such as the states' trout stamps or the federal Farm Bill's conservation titles, the base of support for those programs is significantly broadened. These economic impact surveys provide strong evidence speaking to benefits not only to the streams themselves, but to the communities near which they flow.

Conclusion

In the Driftless Area and elsewhere, natural resource managers and policy-makers at every level should keep in mind the economic impact of angling for wild trout on the communities where coldwater streams are located. Similarly, advocates for wild trout and the restoration of places where they live, or might eventually live, can benefit from pointing out these impacts. While fishing is important in direct terms—aesthetically and recreationally—to anglers who come to visit, wild trout angling can help a community diversify its economy as more streams are restored to health through restoration efforts. The benefit of considering these impacts is not limited to the decisions of a village or county board; it can help members of Congress in their deliberations about the conservation titles of the next federal Farm Bill, or state legislators in a wide range of issues.

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Costs, Benefits, and Challenges of Establishing and Maintaining Real-time Instrumentation that can be used to Inform Management and Educate Stakeholders

Melissa Muradian¹, Rob Van Kirk²

¹Research Associate, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, FAX 208-652-3568, melissa@henrysfork.org.

²Senior Scientist, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, rob@henrysfork.org.

Abstract—Large-scale collection, analysis, and dissemination of natural-resource data have traditionally been undertaken by government agencies. However, agencies such as the U.S. Geological Survey have discontinued monitoring stations due to resource limitations. In response to shrinking agency budgets, nongovernmental organizations (NGOs) must fill the need for real-time information critical to fisheries and water management at fine spatial resolution. Hardware and software are now available to NGOs, but proper investment in staff with capabilities in data analysis and Information Technology is integral to success. In 2014 the Henry's Fork Foundation initiated a water-quality monitoring network spanning the Henrys Fork of the Snake River watershed. The network consists of ten YSI EXO2 multi-parameter sondes that log 15-min data. Such copious data require management software, but existing proprietary choices are expensive and limiting. Furthermore, quantitative analytical skills are required to draw conclusions. We created a full-time position to maintain the network and develop custom software to remove erroneous data points, smooth time series, and perform hypothesis testing. Interns conduct wintertime maintenance, collect water samples, and have developed a web site to inform managers and educate stakeholders. Our network has dramatically increased our scientific and educational capacity, and this success required full investment toward in-house technical and quantitative expertise.

Introduction

Large-scale collection, analysis, and dissemination of natural resource data have traditionally been undertaken by government agencies with a long-range perspective on resource management. Given the physical and chemical connectivity of air, land, and water, agencies such as the National Atmospheric and Oceanic Administration, the U.S. Forest Service, and—most notably—the U.S. Geologic Survey (USGS) collect data and conduct research that is important to tribal, state, and local management of fisheries and water resources. The information these agencies collect and share is becoming even more important as managers wrestle with understanding and predicting short- and long-term effects of climate change, major infrastructure changes (e.g. removal of Elwha Dam), small-scale habitat-restoration projects, and everything in between. However, these agencies are forced to cope with budget cuts when national, state, or local objectives shift. For example, USGS has decommissioned 27 of their stream gauges in Idaho,

Montana, and Wyoming since 2015 due to lack of resources (USGS 2017).

Environmental data collected at both large and local scales are necessary for researchers and managers to meet their objectives. Some research and management questions can be answered only with local-scale data. For example, predicting how an increase in volume of an existing reservoir will impact a resident, endemic fish species requires on-site and proximal data collection. Thus, while it is crucial that large-scale data collection continue, the addition of finer spatial-scale data is needed to monitor baseline status or project-specific watershed-scale effects.

Faced with shrinking agency budgets and authority, nongovernmental organizations (NGOs) must step up to fill the need for real-time information critical to fisheries and water management at fine spatial resolution. Hardware and software are now readily available to NGOs to accomplish this, but proper investment in infrastructure and in staff with capabilities in data analysis and Information Technology is integral to success.

The Henry's Fork Foundation (HFF) is a nonprofit conservation organization that was established in 1984 by a handful of concerned anglers. For over three decades, HFF has consistently focused on restoring trout habitat, improving streamflow management, monitoring trout growth and survival, promoting responsible stewardship of the abundant resources of the Henrys Fork, and securing river access. HFF continues to fulfill its mission through collaboration with multiple stakeholders, including state and federal agencies, and through scientific research. For the first three decades, the overarching goal of HFF's scientific program was restoring or improving trout habitat—including improving streamflow—to promote wild trout populations. After many years of monitoring fish populations, recording river temperature, improving fish passage, and participating in collaborative water management, it became clear that an important tactic was glaringly absent: comprehensive and continuous water-quality monitoring.

The Henry's Fork Foundation initiated a continuous-time water-quality monitoring network in 2014 with plans to span the entire Henrys Fork of the Snake River watershed. The network monitors numerous water-quality parameters, which can be influenced by everything from our own habitat restoration projects to local water-management operations (reservoirs, irrigation diversions, hydroelectric power projects) to seasonal, annual, or climatic hydrometeorological trends. Our water-quality monitoring network gathers detailed and comprehensive baseline information on existing trout habitat, giving us the tools to examine how the location and timing of chronic or acute fluctuations in physical and biochemical states impact wild trout.

Collecting great amounts of data necessitates great data-management responsibility. It is critical to our scientific objectives that our water-quality monitoring network be as comprehensive as possible, which implies a network of several sondes recording many data points as frequently as possible. For example, our network recorded over 1.7 million data points in 2016 alone. Such copious data require management software, but existing proprietary choices are expensive and limiting. What's more, quantitative analytical skills are required to draw conclusions from the data. We created a full-time position to maintain the network, troubleshoot issues, and develop custom software to remove erroneous data points, smooth time series, and perform hypothesis testing. Additional

staff time and internships were required to meet additional demands of wintertime maintenance, collect water samples used to develop turbidity-nutrient and turbidity-suspended sediment relationships, and develop a public web application used to inform managers and educate stakeholders.

Network Design and Installation

The continuous-time water-quality monitoring network was implemented in three phases over 3 years to reduce the upfront cost of equipment and extend the fundraising period. The network currently consists of ten YSI (Yellow Springs, OH) EXO2 multiparameter sondes. These sondes are rated for up to 10 m of water depth and are deployed semi-permanently at fixed locations in the river. We also have an eleventh sonde used for short-term projects or experiments, or to take the place of a sonde in need of repair. These sondes are configured to record every 15 min: temperature, conductivity, turbidity, dissolved oxygen, river depth (or stage), and an index of chlorophyll *a* and cyanobacteria biomass, all of which are important drivers and indicators of trout life-history success, hydrologic processes, and aquatic-ecosystem production in the Henrys Fork.

Spatial Coverage and Experimental Design

One objective behind the design of our water-quality monitoring network is the ability to pair our water-quality data with existing USGS streamflow measurement stations where possible (Figure 1). The active USGS stations in the Henrys Fork watershed collect stage and discharge data. Combining water-quantity data like this with water-quality data provides a complete biophysical context for research and monitoring of fish habitat and fish populations in the Henrys Fork.

The second objective underlying our network's design is the ability to monitor each unique reach of the river. The Henrys Fork, from its source at Big Springs to its confluence with the Teton River, has several distinct reaches that are defined by transitions in hydrologic regime and channel morphology and further differentiated by tributaries, reservoirs, irrigation diversions, and degree of hydrologic connectivity with local and regional aquifers (Benjamin 2000; Bayrd 2006; USBR 2015). There are

two reservoirs located on the main-stem Henrys Fork: Island Park and Ashton reservoirs. We have sondes up- and downstream of both reservoirs to monitor their impact on downstream water quality (Figure 1, sites 1,2,3,7, and 8). We installed two sondes immediately below Island Park Reservoir (Figure 1, sites 2 and 3) because the dam has two discharge points: the original gates installed when the dam was built in the 1930s, and the intake for a hydroelectric project added to the dam in the 1990s. Our sondes monitor water flowing from each of the outlet structures since the intakes are located at different elevations in the water column, and so deliver water of differing quality when the reservoir has stratified, for instance. The 15-km reach that begins upstream of Harriman State Park features the famous wide, flat, and shallow water beloved by dry fly-fisherman. We have a sonde located near the

downstream-most end of this reach (Figure 1, site 5) and upstream of where the river transitions to a reach dominated by rapids and deep canyons. Sites 4 and 6 monitor water quality inside two major tributaries—Buffalo and Warm rivers—just before they join the Henrys Fork. Finally, sites 9 and 10 monitor water quality on the Snake River Plain—an area dominated by irrigated agriculture—where the river is punctuated by numerous points of diversion and interactions with local and regional aquifers. Thus, our sondes are positioned to capture the influence on water quality of each reach and major structure or tributary.

Lessons in Sonde Installation

It is critical to securely fix sonde housings onto existing structure when possible. In our case, existing structure has included docks, bridges, large rhyolite boulders, or basalt canyon walls. Otherwise, the Henrys Fork slowly but surely dismantled our rickety improvisations and threatened to sweep them downstream. We use 100-mm diameter ABS pipe as the sonde housings with 25-mm holes drilled into the bottom 0.6 m of the pipe to allow water to flow over the probe tips, but also to protect the sondes. We secure the bottom and top ends of the pipe to permanent structure along the streambank. For HFF, discreetly fitting sonde housings onto existing structure, rather than building new solid structures on the bank, satisfied a secondary but no less important objective: maintaining the existing aesthetic qualities of the river, which is a component of our mission.

A permanent sonde network that spans an entire river must have nodes that are designed for the unique bank structure and geomorphology of that reach. The river-continuum pattern can be used to aid in design, but nothing takes the place of actually walking the riverbank, sampling the substrate, and observing seasonal changes in flow, stage, and bank conditions. For us, creating site-specific installations, building them, and then inevitably having to rebuild them took up a large amount of personnel time during the first 3 years until we converged on a general installation design that worked well and was reliable. After this, installations were more predictable, using this design template, so site determination and installation time was drastically reduced. There is no substitute for on-the-ground experience; our installations are now more robust because we have improved knowledge after 4 years of confronting unexpected issues specific to the Henrys Fork.

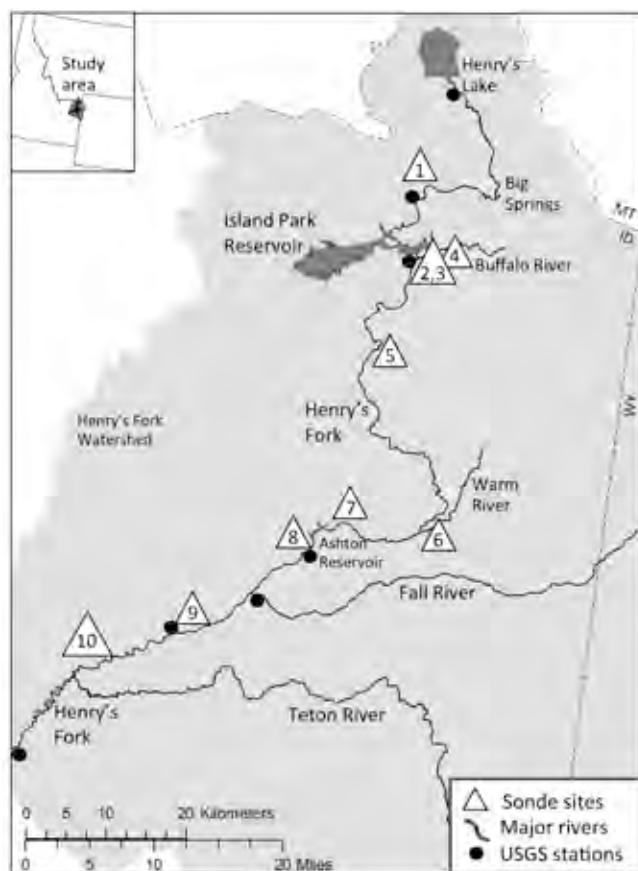


Figure 1. A map of the Henrys Fork watershed (light grey shaded area). The main stem of the Henrys Fork and major tributaries are shown. The continuous-time water-quality monitoring sonde network is denoted by the set of white triangles and black circles denote the relevant USGS streamflow gauges in the watershed.

Maintaining a Year-round Sonde Network

The sondes themselves require regular maintenance during deployments, and we manually download the recorded data every 2 weeks. The probes must be calibrated every 2 months to maintain instrument accuracy. All of our sondes are currently powered by D-cell batteries, which need to be changed every 3 months under the best conditions. Memory storage in the EXO2 is capable of holding multiple years worth of 15-min data, but we have found that an optimal schedule for downloading data from the sondes is every 2 weeks; this minimizes data loss if something goes wrong with a sonde or probes, while also conserving personnel time. One of the biggest sources of data corruption is when uprooted aquatic plants drifting downstream get caught around and inside of the sonde housings. During peak growing season, we regularly remove hundreds of pounds of uprooted plants from our housings. Plants stuck or waving in front of the sensors will lead to erroneously high turbidity and chlorophyll *a* readings. Dogs and people recreating in the river may kick up sediment in front of the sondes while they are making a measurement, which can lead to false readings in turbidity, chlorophyll *a*, or cyanobacteria.

During the winter months there are fewer variables that can corrupt data, but wintertime fieldwork in the high plains of Eastern Idaho is time consuming. We are able to keep four of our sondes in the river through the winter months since the river does not freeze in those places. The EXO2 can keep recording if fully submerged below ice cover, but none of our sites that freeze have high enough winter flow to submerge a sonde completely beneath the ice cover.

A secondary installation may be necessary to accommodate data collection at a site with large changes in seasonal flows and stage. This was required at a site where the bank structure prevented an installation that was long enough to ensure the sonde end was under water during periods of low river stage while the top was still accessible during periods of high river stage. For instance, at sonde site 2 (Figure 1) we installed two permanent housings in close proximity but at differing heights, and we simply transfer that site's sonde from one to the other when necessary.

Data Processing and Visualization

Data management software is required to make such copious amounts of raw data useful, but existing proprietary choices are expensive and limiting. The HFF science team already uses the statistical programming environment R for statistical analyses, hydrologic modeling, and generation of graphics. R is free, open-source, and allows us to have complete control over formatting preferences, correction algorithms, and visualization methods.

Some data formatting preferences can be subjective, but formatting your data is a necessary first step of any kind of quantitative research. We developed an algorithm that reads a recently downloaded data file, runs a routine to automatically remove erroneous data points, and then appends the cleaned data to a master file of all data collected from that sonde site. To create a continuous time series from our sonde data this algorithm also keeps track of a continuous sequence of 15-min intervals, and fills in blank rows if there is a gap in time between files. Finally, this formatting algorithm also derives river stage local to each sonde by adjusting total pressure recorded by the sonde for atmospheric pressure recorded by a central network barometer (Figure 2, bottom panel). This stage data can be paired with the USGS stage data, where possible, to reciprocally validate stage records. Our stage data have been used by USGS to generate streamflow records during brief times when one of its stage recorders malfunctioned and also to identify control sensitivity in Ashton Reservoir's run-of-river hydropower facility.

The cleaning routine mentioned above, which removes erroneous data points, is based on anomalies in the conductivity data and the turbidity data. We define erroneous data points as readings that the sonde made that do not indicate true overall water quality at that time either because something—or someone—obstructed a sensor or because the sonde was out of the water. Anomalies due to the former were discussed in the last section and need to be removed from the turbidity record and sometimes from the chlorophyll *a* or cyanobacteria record. On the other hand, dry data points need to be removed from the record for all parameters and are most easily and reliably discerned from the conductivity record. In either case, these anomalies introduce suddenly very high or very low values into an otherwise fairly smooth time series.

The cleaning algorithm applied to the turbidity and conductivity algorithm works the same way, using different tolerances appropriate for each parameter. First, the algorithm calculates the difference between a value and that of the previous time step. Based on analysis of 4 years of data, we have found that actual turbidity values in the river should not differ by more than 2 Formazin Nephelometric Units (FNU) over any 15-min interval. Thus, these data points are removed (Figure 3, top panel). This leaves behind a smoother series in many places, but erroneous values can remain. For example, if several plant stems and leaves are randomly waving in front of the turbidity sensor, there may be individual readings that capture some subset of plant parts, while others capture all plant parts. These readings may differ from each other by less than 2 FNU, our tolerance, and would remain in the series (Figure 3, middle panel).

A forward and backward moving-range routine effectively removes remaining errors in the data. The

forward moving range works by saving the pair of minimum and maximum values over a window of time, and then as the window moves forward by 1 data point, a new pair of minimum and maximum is calculated. All pairs are saved and those pairs that differ from the last pair by more than a specified amount are flagged. The backward moving-range is calculated the same way and those flagged pairs that line up in time with those from the forward-moving range identify periods where the data abruptly differs from the surrounding values (Figure 3, bottom panel). How abrupt of a change is too abrupt? This question is central to the effectiveness of the algorithm since we expect these parameters to change, and their rates of change can differ, but from studying our data we have determined the cut-off rate for changes that most-likely do not relate to true fluctuations in the parameter.

Clear visualizations of our data are key to monitoring and research. We review the 6 primary parameters collected by our water-quality monitoring

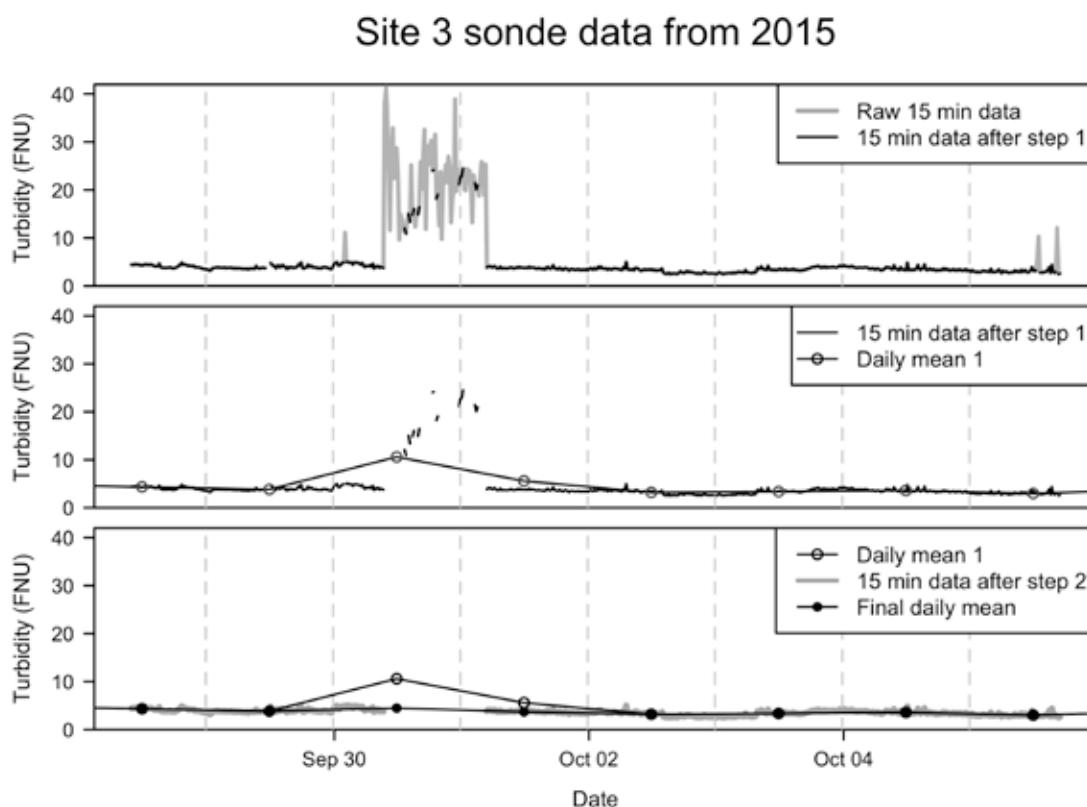


Figure 3. A period of turbidity data where the first step in the cleaning algorithm leaves behind scattered erroneous data (top panel). The moving-range algorithm ultimately resolves this period of troublesome data (bottom panel). Vertical dashed lines separate days. Daily averages are shown to demonstrate how the full algorithm improves interpolation across missing or removed data.

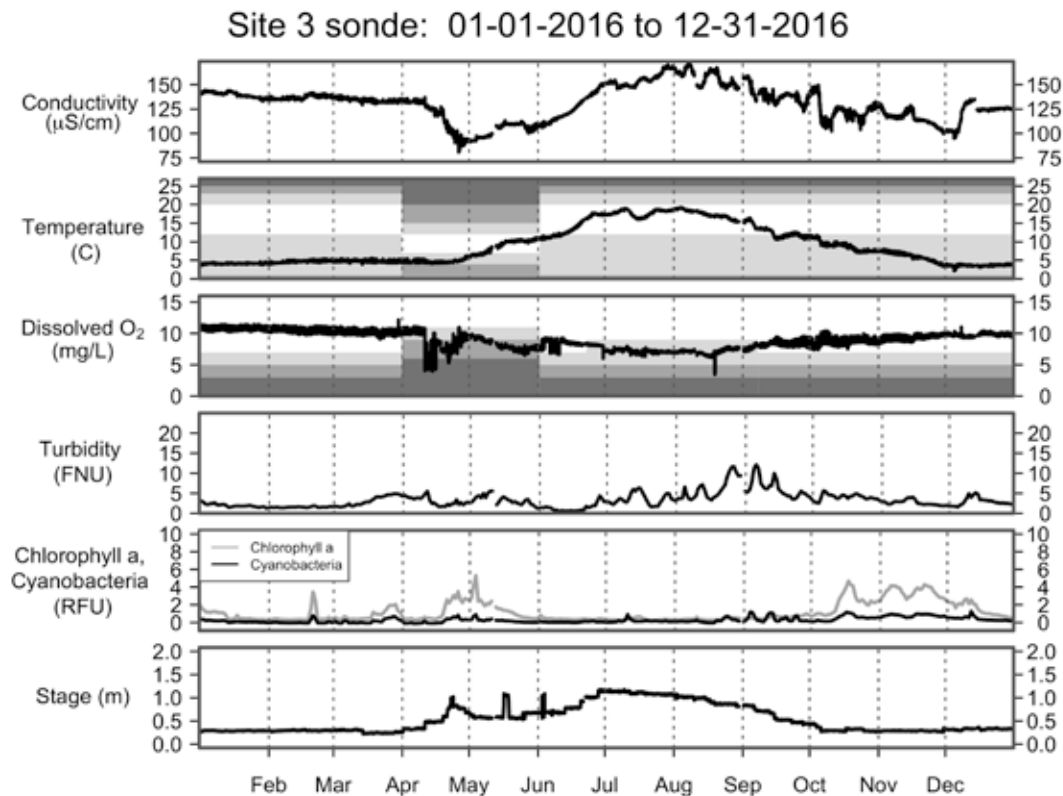


Figure 2. A complete year of data from our sonde at site 3. Conductivity, temperature, dissolved oxygen, and stage are 15-min data. Turbidity, chlorophyll a, and cyanobacteria are daily averages produced using our automatic cleaning algorithm. Color-based thresholds of habitat quality for Rainbow Trout *Oncorhynchus mykiss* are shown in the temperature and dissolved oxygen panels (Raleigh et al. 1984). Darkest grey signifies lethal conditions; middle grey is stressful; lightest grey is suboptimal; and white is optimal.

network side-by-side to observe a detailed summary of water quality conditions at any given time (Figure 2). Comparing information contained in each parameter to the others allows us to reliably discern issues with equipment, identify trends in overall water quality, and hypothesize explanations for short-term variations. We also add color-based thresholds of Rainbow Trout habitat quality, from literature review, to the temperature and dissolved oxygen panels (Raleigh et al. 1984). Darkest grey signifies lethal conditions; middle grey is stressful; lightest grey is sub-optimal; and white is optimal. Black and white images are produced for the current publication, but we use red, yellow, and shades of green to represent lethal-to-optimal conditions and these thresholds allow for immediate and intuitive interpretation of the current water quality and how it relates to fish health.

Data Dissemination

We set out to develop a public data web site and companion mobile application to provide freely accessible archived and real-time water quality data that could be used to easily share data across partner organizations, inform managers, and educate stakeholders. The first step was enabling real-time data transmission from a sonde to a server in our office. Research into existing technologies (hardware and software) resulted in a data stream involving YSI EXO adapter, Campbell Scientific (Logan, UT) data logger and cell modem, Verizon cell service, Amazon database manager, and R Shiny web application. To get these data in the hands of many stakeholders, a mobile application streamlines navigation to the web site. Additional benefits of enabling real-time data transmission include reduction in field time—allowing

our science team to spend more time on research and analysis—and real-time in-house monitoring of monitoring equipment status and data quality.

Analyses

To date, we have used our sonde data to statistically analyze daily cycles in streamflow caused by macrophyte respiration, dependence of turbidity and dissolved oxygen concentrations on management of Island Park Reservoir, sediment and nutrient budgets for the reservoir and adjacent river reaches, and dependence of water temperature on Island Park Reservoir hydrology. We have presented results at scientific conferences, at regular meetings of the Henry's Fork Watershed Council, and to our members and stakeholders via multiple media. Our results have also been used to inform a study on the role of macrophytes in creating trout habitat in the Harriman State Park reach of the river (Kuzniar et al. 2016) and are regularly incorporated into multi-stakeholder discussions of reservoir and hydropower management on the river.

Conclusions

State-of-the-art technologies to implement real-time water-quality monitoring and data transmission are affordable, reliable, and readily accessible to NGOs. After a few years of time spent by highly skilled staff planning, troubleshooting issues, and developing analytical tools, our monitoring network has dramatically increased our current and future

scientific and educational capacity. However, without full investment of in-house technical and quantitative expertise such an ambitious monitoring program can easily become burdensome or underutilized, leading to a low return on the cost and field time required to set up and maintain such a monitoring network.

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Angler Perception of Fishing Experience in a Highly Technical Catch-and-Release Fishery: How Closely does Perception Align with Biological Reality?

Jamie Laatsch¹, Rob Van Kirk², Christina Morrisett³, Kaitlyn Manishin⁴, Jim DeRito⁵

¹Conservation and Outreach Coordinator, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, FAX 208-652-3568, jamie@henrysfork.org.

²Senior Scientist, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, rob@henrysfork.org.

³Research Assistant, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, christina.morrisett@gmail.com. Present address: School of Aquatic and Fisheries Sciences, University of Washington, Seattle, WA, 98195.

⁴Statistical Consultant, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, kmanishin@gmail.com. Present address: College of Fisheries and Ocean Sciences, University of Alaska, Fairbanks, AK 99775.

⁵Fisheries Restoration Coordinator, Trout Unlimited, 44 W. Spring Creek Parkway, Providence, UT 84332, 208-360-6165, JDeRito@tu.org.

Abstract—Recreational fisheries management, and indeed natural resource management as a whole, faces increasingly complex challenges as resource-user attitudes and expectations clash with the realities of a changing climate and increasing demands on water resources. We interviewed anglers in a highly specialized catch-and-release trout fishery in southeastern Idaho in 2008 and 2014, followed by a separate, online survey in 2016, to better understand angler attitudes and assess coherence between perceived angling conditions and observed environmental conditions, such as size and quantity of fish, macroinvertebrate abundance, and water quality. Anglers were contacted at fishing access points and asked about their experience level, trip characteristics, expectations, satisfaction with specific attributes of the fishery, and satisfaction with overall fishing quality. Results indicate that angler satisfaction is more closely tied to perceived aesthetics of a given fishing trip than to measurable ecological variables. Furthermore, a higher fraction of 2016 survey respondents cited agricultural water use as a contributor to poor fishing conditions than acknowledged the effect of climatic factors. This poses a great challenge for recreational fisheries managers, but also indicates that both research into angler perceptions and values, and angler education could be crucial strategies for successful future management of the resource.

Introduction

In a watershed where the agricultural economy overshadows the recreational fishing economy roughly 50 to 1 and a changing climate has brought both a 3°C increase in average springtime temperatures over the past four decades and a record 4-year drought from 2012 to 2016, the challenges of managing a river for healthy ecological function are inherently complex. The challenge is then even greater for a conservation nonprofit organization whose members expect steps to be taken to reclaim the idyllic conditions that made the Henrys Fork a world-famous wild trout fishery and international fly-fishing destination in the 1970s, year-after-year. As with most natural resource management issues today, the management of this recreational fishery is made more complicated by factors like

climate change and human perceptions and values. However, how do managers adapt when those human perceptions do not align with the biological reality of the river according to observed environmental conditions via scientific data?

In 2016, fishing conditions on the Henrys Fork of the Snake River were poor by any measure, and the Henry's Fork Foundation (HFF), a conservation nonprofit focused on the conservation and protection of this river, began hearing a great deal of concern from anglers regarding the health of the river. Thankfully, HFF's science program, centered on water quality and quantity, already had a robust monitoring program in place. Unfortunately, HFF staff began to notice a disconnect between the concerns of anglers and the data gathered from the monitoring program, especially in the Henrys Fork's most iconic reach,

the Harriman State Park or “Harriman” reach. So, in July of 2016, HFF set out to better understand how closely angler perceptions aligned with the observed environmental conditions in the river. Using two key data sets, one collected in 2008 and 2014 as part of an angler attitudes survey, and another collected in 2016 as part of an online survey of angler perceptions, this paper addresses the following questions: (1) What do anglers value about the Henrys Fork? (2) How does angler satisfaction with angling conditions vary across years? (3) What factors predict angler satisfaction? (4) How do angler perceptions align with or stray from the observed environmental conditions in the river? We hypothesized that angler satisfaction would be correlated with biological factors that directly relate to the fishery (e.g., fish abundance, macroinvertebrate assemblage structure, and habitat conditions).

The purpose of this multi-year study is to better understand angler perceptions and assess coherence between perceived angling conditions and observed environmental conditions in the Harriman State Park reach of the Henrys Fork. The 2008 and 2014 surveys aimed to identify predictors for angler satisfaction and compare changes in angler characteristics and perceptions over time. Then, the 2016 survey was designed to better understand the potential disconnect between angler perceptions and observed environmental conditions, after a particularly difficult year for fishing conditions and angler satisfaction, so that HFF could improve communications with anglers and hopefully improve overall angler satisfaction. HFF also had the objective of assessing how various factors (i.e., angler characteristics or values) might impact angler perceptions.

Study Area

Harriman State Park is a 45-km² wildlife refuge located within the Greater Yellowstone Ecosystem on the Henrys Fork, a groundwater-fed tributary to the Snake River in southeastern Idaho, USA. Originally known as the Railroad Ranch, the Harriman family gifted the cattle ranch to the people of Idaho in 1977, and it became Idaho’s first state park in 1982. The 13-km Harriman reach of the 180-km Henrys Fork is famous for its insect hatches and large, wild Rainbow Trout *Oncorhynchus mykiss* (Lawson 2012; McDaniel 2012). This reach is characterized by a wide (mean 160 m; Henry 2010), shallow channel (0.4–0.8 m, depending on flow and macrophyte abundance; Kuzniar 2016) with slow water velocities

(0.2–0.5 m/s during angling season, depending on flow and macrophyte abundance; Kuzniar 2016), gravel substrate, and minimal riparian vegetation (Jankovsky-Jones and Bezzerides 2000; NewFields 2012; Kuzniar 2016). Seasonal growth and senescence of macrophytes greatly influence trout habitat availability and ecological processes in the Harriman reach (Vinson et al. 1992; Van Kirk and Martin 2000; Kuzniar 2016). River flow through the Harriman reach is dependent on regulated reservoir release from Island Park Dam, located 8 km upstream. Under the regulated flow regime, in which water is stored and released for irrigation, minimum flow averages 8.8 m³/s and occurs in November, and peak flow averages 400 m³/s and occurs in July (Benjamin and Van Kirk 1999). The Buffalo River, a major tributary between the dam and the Harriman reach, contributes around 6 m³/s of unregulated flow year-round.

Prior to 1994, the trout population in the Harriman reach and adjacent reaches was partially supported by hatchery fish, via downstream migration of fish stocked into Island Park Reservoir, and, prior to 1978, via direct stocking (Van Kirk and Gamblin 2000). The current, wild-trout population is limited by overwinter survival of juvenile fish, which is directly related to winter flow release at Island Park Dam (Gregory 2000; Mitro et al. 2003; Schoby et al. 2010).

Under a condition of the Harriman’s gift, the Harriman fishery has been managed under catch-and-release, fly-fishing-only regulations since the 1970s and is a popular destination for domestic and international dry-fly anglers eager for the challenge of presenting insect-specific imitations to selectively feeding trout on flat water (Van Kirk and Griffin 1997; Van Kirk and Gamblin 2000). The fishery is legally open from June 15 to November 30, but low water, cold weather, and lack of insect activity end the effective fishing season in mid-October. Angling effort is highest from opening day through mid-July, when the most popular insect hatches occur. A typical angling trip is 4 to 6 h long, from late morning through early afternoon, when insect activity is greatest. Early in the season, some anglers also fish for a few hours in the evening. During autumn, the typical 4- to 6-h trip occurs during the afternoon. The Harriman reach is the centerpiece of recreational trout fisheries that support 170,000–220,000 angler-days of effort and US\$51–US\$60 million in annual economic activity in the Henrys Fork watershed (Loomis 2006; Grunder et al. 2008).

Methods

2008 and 2014 Angler Attitudes Survey

A clerk interviewed anglers at the completion of their daily fishing trip, usually in the afternoon, at each of the seven fishing access points in the Harriman reach. Clerks were stationed at one or more of the seven access points for 4 to 6 h every day from June 15 to mid-October in 2008 and 2014. Survey effort was distributed roughly in proportion to spatial and temporal distribution of angler effort, both within and across days. Clerks attempted to interview every angler who completed a trip while the clerk was on duty, but during busy times, clerks were unable to interview all anglers. Upon first encounter, each angler was given a bird band with an ID number so that individual anglers could be identified throughout the fishing season. In the case of anglers who were interviewed more than once within the year, one of their interviews was selected at random for inclusion in the statistical analysis. Sample size was $n = 972$ unique-angler interviews, 616 from 2008 and 356 from 2014. Of these, 82 were at least the second interview with an angler who was interviewed more than once within a year. Not all survey respondents answered every question, so sample sizes varied across different statistical analyses.

Survey questions were motivated by informal conversations with anglers who had expressed dissatisfaction with the fishery prior to 2008. The survey instrument contained a group of general questions that characterized the angler's experience level and primary residence location. Experience level was quantified by years of experience with the Harriman fishery, number of days fished per year both in the Harriman reach and on other waters, and number of days fished in the Harriman reach during the current season up to and including the interview day. Anglers were then asked how many of their Harriman fishing days were guided and how many fish they caught on the interview day.

Anglers were asked to rate the quality of the day's fishing experience (excellent, good, fair, poor). They were then asked to rate their expectations for each of seven aspects of the Harriman angling experience: number of fish caught, size of fish caught, number of other anglers on the river, and aesthetic qualities of the river (e.g., scenery), number of opportunities to fish to

rising fish, quality of the insect hatches, and condition of fish habitat. These questions were asked in the form "how important is each of the following factors in determining whether you have a high-quality fishing experience in the Harriman reach?" Importance was measured on a 10-point scale (1 = not at all important; 10 = very important).

2016 Survey

A nine-question online survey was developed and distributed to anglers via email and social media in 2016. The electronic survey included questions about fishing conditions, angler understanding of river hydrology, and angler characteristics. More specifically, the survey began by asking respondents to rate fishing conditions on a scale of 1-10 (1 = very bad; 10 = very good) and select which factors they believe impacted fishing conditions for two separate sections of the river from June to August 2016. The first section spanned Island Park Dam to Riverside Campground, which contains the most popular reaches of the river, including the Harriman reach. The second section simply included all other sections of the Henrys Fork. The factors respondents had to choose from included high flows out of Island Park Dam, low flows out of Island Park Dam, cyanobacteria blooms, warm summer temperatures, lack of rainfall, wind, high turbidity, high number of other anglers on the river, and low number of other anglers on the river.

Respondents were then asked a series of questions pertaining to any and all sections of the Henrys Fork; however, only respondents who had fished the Harriman reach were included in final analysis. First, they were asked to rank a series of potential factors based on how significantly they believed each factor impacted how much water was delivered from Island Park Dam during the summer. These factors were snowpack, spring/summer rain, river base flows, and irrigation demand. They were also asked a series of questions to get a better sense of their behaviors and values as an angler. These included questions about the timing and frequency of their fishing trips on the Henrys Fork as well as what factors they value most about the river (i.e., world-class wild trout fishery, fish per mile, big fish, hatches, peace and quiet, scenery, proximity to National Parks). Finally, respondents were provided space for comments, feedback, or questions at the end of the survey. In total, 103 anglers responded to the survey.

We acknowledge possible bias in the informal online survey because respondents were not randomly selected. In particular, we suspect that anglers who responded were those who already felt most strongly about fishing condition or were members of HFF who are predisposed to care strongly about the quality of the fishery.

Data Analysis

We used standard descriptive statistics to summarize angler characteristics and values. For this summary we pooled data from 2008 and 2014 to represent a single angler population and considered the 2016 population to be potentially different. Because questions regarding angler values differed slightly between the 2008/2014 and the 2016 surveys, we grouped them by attributes that most closely aligned between the two surveys. Furthermore, for consistency in comparison between the two surveys, we reported percent of respondents rating a particular value above average (6-10 on the 10-point scale) in the 2008/2014 survey and reported total percentage of respondents who selected the particular attribute as important in the 2016 online survey.

We compared angler satisfaction among the three distinct years 2008, 2014, and 2016. We first converted angler satisfaction to a binary response: 0 = below average (“fair” or “poor” in 2008 and 2014; 1-5 on the 10-point scale in 2016) and 1 = above average. We then compared fraction of respondents rating the fishing above average across years with a generalized linear model using the binomial distribution and logit link (glm function in R statistical programming language). We used the likelihood ratio test to compare a model in which satisfaction differed across years with a model in which satisfaction was assumed to be constant across years (null hypothesis). We used analysis of variance on the raw 10-point satisfaction ratings to compare angler satisfaction in 2016 across seasons.

We correlated angler satisfaction with environmental variables using ranks. Because the observational unit for the environmental variables was year and we had only 3 years of data, no formal statistical inference was possible. However, we considered there to be a correlation between an environmental variable and mean angler satisfaction if the rank of the environmental variable across years exactly matched (positive correlation) or was exactly opposite (negative correlation) the rank of

angler satisfaction. Specifically, angler satisfaction was highest in 2014 (rank = 1), intermediate in 2008 (rank = 2), and lowest in 2016 (rank = 3). Thus, for example, an environmental variable that was highest in 2014, intermediate in 2008 and lowest in 2016 would be positively correlated with angler satisfaction. Environmental variables used in this analysis were trout abundance, mean trout size, macrophyte cover, two indices of macroinvertebrate assemblage structure, mean flow out of Island Park Dam over the fishing season, and mean daily maximum temperature over the fishing season.

To analyze the dependence of summer flow out of Island Park Dam on the hydrologic factors listed in the 2016 online survey, we conducted linear regression analysis of mean June 15 – August 15 outflow from the dam as a function of April-1 snow water equivalent, June-August precipitation, stream base flow (October-March natural streamflow in the upper Henrys Fork watershed), and watershed-total irrigation diversion. The sample for this analysis was water years 1988-2016. All possible models using these four predictors were fit, and model-averaged R^2 values (percent of sum-of-squares explained) were calculated for each predictor. Order of influence on high summertime delivery was determined by the R^2 values.

Results

Profile of Harriman Anglers

Results from the 2008 and 2014 surveys showed that most Harriman anglers reside in Idaho, Utah, or California (ID = 22%, UT = 16%, CA = 14%). The median Harriman angler began fishing the Henrys Fork in 1995, fishes the Harriman reach 6 d per year, fishes other sections of the Henrys Fork 3 d per year, fishes 45 d per year in all locations, does not fish with a guide, and catches 0 fish in the Harriman reach per day (Table 1).

In both surveys, anglers indicated that they valued more visual or aesthetic qualities (rising fish, hatches, and scenery) over qualities like number of fish caught and size of fish caught (Table 2).

Angler Satisfaction and Environmental Conditions

Angler satisfaction was significantly lower in 2016 than in either 2008 or 2014 (likelihood ratio test, $\chi^2 = 31.2$, $df = 2$, $P < 0.001$; Figure 1). The only

Table 1. Characteristics of Harriman State Park Anglers. Except where indicated, data are pooled from the 2008 and 2014 surveys.

Characteristic	Min.	Median	Mean	Max.	% reporting 0
Year first fished Harriman reach	1939	1995	1994	2014	NA
Year first fished Henrys Fork (2016 data)	< 1960	1991	~1991	2015	NA
Days fish Harriman reach/year	0	6	12.2	120	1.9%
Days/year fish other Henrys Fork reaches	0	3	9.7	250	25.6%
Days/year fish all locations	0	45	60	325	1.1%
Days fished Harriman reach with guide	0	0	0.12	20	94.9%
Fish caught in Harriman reach/day	0	0	1.0	18	60.6%

Table 2. Fishing-experience values of Harriman State Park anglers. Attributes are listed as they were presented to anglers in the surveys.

Attribute (2008/2014)	% of 2008/2014 responses ranking attribute ≥ 6 on 10-point scale	Attribute (2016)	% of 2016 respondents selecting the attribute as important
Rising fish	90.3%	World-class fishery	85.4%
Hatches	93.9%	Hatches	79.8%
Aesthetics	96.4%	Scenery	76.4%
Size of fish caught	66.0%	Big fish	75.3%
Other anglers	61.0%	Peace and quiet	60.7%
No. fish caught	31.4%	Fish per mile	36.0%

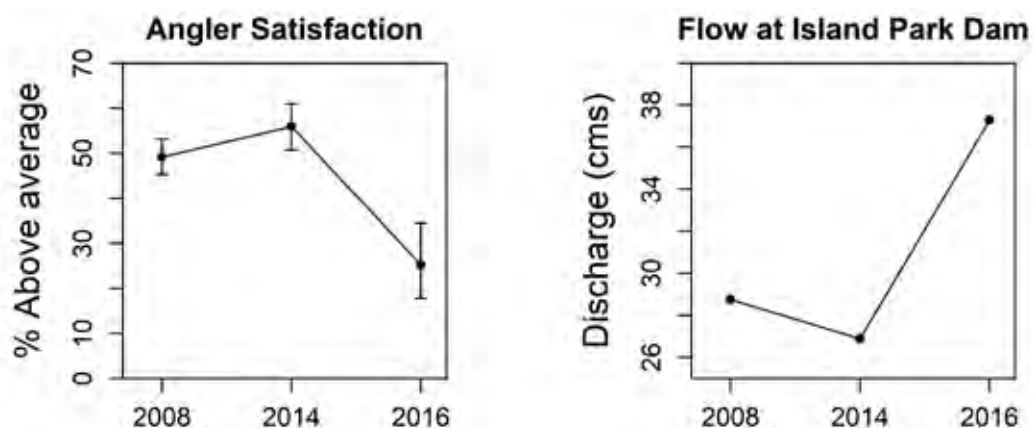


Figure 1. Left panel: angler satisfaction across the three survey years. Error bars indicate 95% confidence intervals. Right panel: mean June 15 – August 15 flow at Island Park Dam.

environmental variable with the same (or opposite) rank-order as angler satisfaction was flow out of Island Park Dam from June 15 to August 15. Higher flows correlated with lower angler satisfaction (Figure 1). Anglers confirmed this in 2016 when a majority of anglers selected “high flows out of Island Park Dam” as a factor impacting fishing conditions in the Harriman reach and selected this factor twice as often as any other factor. Furthermore, in 2016, anglers were significantly less satisfied with the Harriman reach angling experience in June and July, when flows were highest, than in other months of the year ($F = 4.42$, $df_1 = 2$, $df_2 = 90$, $P = 0.015$).

Angler Perception vs Environmental Conditions

When asked which factors impacted how much water is in the river during the summer months, effectively which factors were to blame for high summer flows, 90% of anglers selected irrigation demand downstream. Fewer than 70% of anglers selected this year’s snowpack, 60% selected spring and summer rains, and 51% selected base flows. In other words, a higher fraction cited agricultural water use as a contributor to poor fishing conditions than those who acknowledged the effect of climatic factors. On the other hand, the regression analysis showed that climatic factors, specifically this year’s snowpack ($R^2 = 0.28$) and spring-summer rain ($R^2 = 0.19$), were more important than irrigation diversion ($R^2 = 0.11$) in explaining year-to-year variability in summer flow at Island Park Dam.

Discussion

Anglers indicate that they value the visual qualities of the fishing experience (hatches, scenery, and rising fish) over qualities like number of fish caught and size of fish. However, among a number of environmental and biological variables, only summer-time flow below Island Park Dam was correlated in either direction with angler satisfaction. In this case, lower satisfaction corresponded with higher flows. There are certain visual qualities river anglers are likely to notice like high flows, and the turbid or “dirty” water that comes with those high flows, and whether or not insects are hatching or fish are rising. These visual factors are largely unrelated to fish abundance, macroinvertebrate structure, or habitat conditions, but do impact fishing conditions.

This study has revealed two key points of disconnection between angler perceptions and the biological reality of measurable environmental conditions in the river. First, based on what they can observe while fishing—specifically high flows, poor hatches, and few rising fish—anglers have inferred that undesirable aesthetic qualities equate to negative biological conditions. Second, anglers tend to disproportionately blame irrigation demand for these high flows and poor angling conditions, more than the climate factors that indeed play a bigger role in determining those flows. It makes sense that anglers would assume irrigation plays a larger role because this system was built to store, deliver, and divert irrigation water, but misperceptions about the hydrologic system and climate present a major challenge for local natural resource managers.

As climate continues to change at both global and watershed scales, it becomes more and more critical that anglers and recreational fisheries managers share a common understanding of the environmental realities of the river. A disconnect between angler perception and observed environmental conditions will be a major barrier to future management and must be remedied to ensure effective future management strategies. Also, if anglers are to have real input into management decision-making processes related to public resources, they deserve to be equipped with accurate and relevant information. Nonprofit organizations, as well as state and federal land management agencies, can play a role in closing that information gap by improving communication strategies.

If angler satisfaction is more closely tied to perceived aesthetics of a given fishing trip than to measurable environmental variables, and a majority of anglers misunderstand the relationship between agricultural and climate factors to fishing conditions, then conservation groups need to close this information gap by facilitating dialogue with anglers to better understand their perceptions, and by facilitating angler education on hydrology, climate, water law, and irrigation practices. Recreational fisheries managers will also need assistance in encouraging anglers to better adapt to changing angling conditions. Agricultural producers and others are already adapting to changing climatic conditions, and anglers will need to follow suit if they hope to maximize their satisfaction with their angling experience.

Lastly, this study suggests that our operating hypothesis was wrong. As biologists, we expect angler

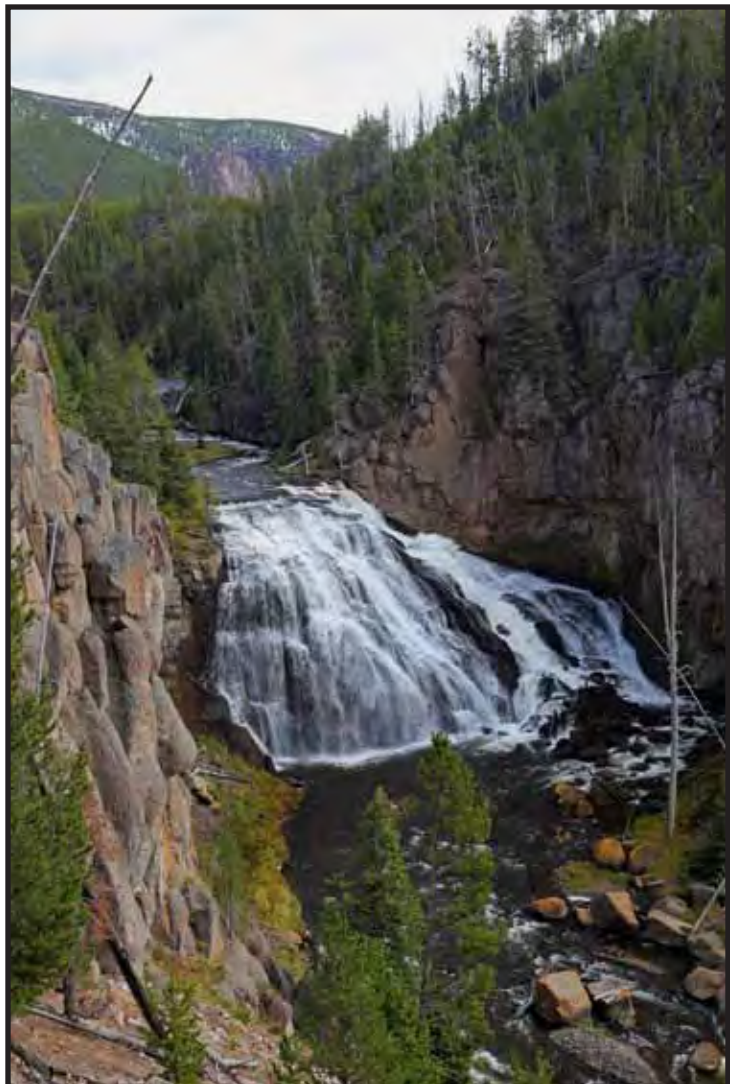
satisfaction to respond to fish size, fish populations, macroinvertebrate assemblages, and habitat quality. Furthermore, as a conservation organization, we have historically focused our efforts on conserving large-scale ecological processes that maintain high fish populations, fish growth rates, habitat conditions, and overall aquatic health, and we monitor traditional indicators such as fish abundance, macrophyte cover, and macroinvertebrate assemblages. However, the results of this study suggest that anglers are responding more to what they see on the river during a given day of fishing. We have been working under the assumption that we should influence the management of the river to the benefit of fish, macroinvertebrate populations, and habitat quality, but if that is not what drives angler satisfaction, should we be focusing on something else?

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Stakeholders Participation in Conservation of Brown Trout Stocks in Serbia

Predrag Simonović^{1,2,§}, Ana Tošić¹, Dubravka Škraba Jurlina¹, Jelena Čanak Atlagić², Vera Nikolić¹

¹University of Belgrade, Faculty of Biology, Belgrade, Serbia

²University of Belgrade, Institute for Biological Research “Siniša Stanković”, Belgrade, Serbia

Abstract—Native Brown Trout *Salmo trutta* reveal remarkable diversity in Serbia, with eight exclusive and a couple of more widely spread mtDNA haplotypes. Insufficiently controlled stocking is a main threat to that, as well as to the amenity value of wild Brown Trout. Native or wild Brown Trout stocks deteriorated as a consequence of stocking with inappropriate strains. Three nonnative trout species and three Brown Trout strains stocked into streams of Serbia were classified as medium-high to high risk of being invasive. Alien Brown Trout strains crossbreed with, and incorporate into the native Brown Trout stocks. Fly fishermen in Serbia greatly differ in attitudes towards fishery policy, management, and conservation of native Brown Trout stocks. They are opposed to conservation measures that may affect angling quality. Though declaratively being conservation-friendly, the majority of anglers consider stocking more efficient than Catch-and-Release (C&R) regulations. Despite good conservational and fishery effects of C&R, increases in license sales from fishermen presses trout farmers and fishery managers to stock, regardless of adverse effects on native Brown Trout stocks. Subsidiaries to farmers and grants awarded to fishery managers could promote stocking of native strains and conservation of native stocks.

Introduction

The Brown Trout *Salmo trutta* L., 1758 is the most widespread fish species native to Europe and adjacent regions in Northern Africa, Asia Minor and Near East (MacCrimmon and Marshal 1968; Behnke 1986; Elliott 1994). Owing to its broad dispersal, great plasticity in life history traits and long-term existence, the Brown Trout is a very polymorphic taxon. That resulted in description of over 25 nominal Brown Trout species (Kottelat 1997). Using the variability that occurs in the Control Region of mitochondrial DNA as a molecular marker, Bernatchez et al. (1992), Suárez et al. (2001), and Bardakçi et al. (2006) described six lineages of Brown Trout: Atlantic (At), Danubian (Da), Adriatic (Ad), Mediterranean (Me), *marmoratus* (MA), Duero (DU) and Tigris (TI). Initially, the greatest diversity (14 haplotypes) was ascertained in the Da lineage, which prompted Bernatchez (2001) and Cortey et al. (2004) to consider it ancestral for all other lineages of Brown Trout. Marić et al. (2006) described three new Da haplotypes from streams in southeastern Serbia: Da*Vl (GenBank Accession Number #DQ318123), Da*Dž (#DQ318124) and Da*Vr (#DQ318125) as well as four new Ad haplotypes in southern Serbia: Ad*Pe (#DQ318126), Ad*Ti (#DQ318127), Ad*Bož (#DQ318128) and Ad*Prz (#DQ318129). Tošić et

al. (2014) discovered another Da haplotype from eastern Serbia, Da23c (#KC630984). This diversity of haplotypes emphasizes the importance of Serbia in the conservation of Brown Trout (Figure 1).

Popularity of Brown Trout comes from their attractiveness, especially for fly fishing, which was practiced from ancient Greeks, to medieval Europeans, to North Americans and Westerners in the industrial period, and to modern anglers throughout the world (Herd 2002). It was stocked in 42 countries worldwide (Welcomme 1992), being the 13th of the most widely introduced species (Fausch 2007). Brown Trout originating from the Eastern Atlantic slopes were initially domesticated for stocking in 1748 in Westphalia in Germany (Leitritz and Lewis 1980). Their ease of rearing facilitated stocking throughout the World. Even in our time, the Brown Trout introduced into the USA in 19th century is known as “German trout”, alluding to the origin of imported brood stock from Brown Trout hatcheries in Alsace and Baden-Württemberg (Müller 1956). The impact of stocking on the native trout in North America was great in both east (i.e., Atlantic) and west (i.e., Pacific) slopes (Behnke 2007), and affected especially the Westslope Cutthroat Trout *Oncorhynchus clarkii* (Richardson, 1836). Introduced Brown Trout also posed strong adverse effects to other native, non-trout fishes elsewhere (McIntosh et al. 1994; McIntosh and Townsend 1995; Glova 2003), which has labeled Brown Trout as one of the world’s most invasive

§Corresponding Author e mail: pedja@bio.bg.ac.rs¹

alien fish species (Lowe et al. 2004). Invasive effects of alien strains of Brown Trout and trout species on native fish in trout streams of Serbia were estimated by Simonović et al. (2015a), revealing the very high risk of invasiveness from Rainbow Trout *Oncorhynchus mykiss* (Walbaum), but also from domesticated Brown Trout of At1 and Da2 haplotypes, nonnative for the Balkans. In contrast to low likelihood of massive Rainbow Trout introduction which would be illegal, there are no legal obstacles for stocking of attractive Brown Trout fisheries with Brown Trout, if that is justified in the fishery's Management Plan. The increasing impact of recreational fishing to wild Brown Trout by insufficiently controlled stocking of trout fisheries and by introducing the nonnative Brown Trout strains raise the threat to their original diversity. Therefore, there is a strong necessity for altering Brown Trout fishery management to minimize the possible risk that stocking might pose.

The regulatory Catch-and-Release (C&R) fishing regime is used as a mandatory protective measure that provides a sustainable fishery by decreasing fishing pressure on fish stocks. C&R became widely applied in managing recreational fisheries since 1970 (Barnhart 1989). It was combined with the use of barbless hooks to ensure low mortality after repeated hooking events (Jenkins 2003; Pope et al. 2007) and encouraged as a voluntary sportsmanship. Policansky (2007) reported that voluntary C&R was either not widely adopted (e.g., in Norway), or even forbidden (e.g., in Germany). Arlinghaus et al. (2007) stated that long-term practicing of the voluntary C&R can even be harmful, leading to overcrowding, stunted growth, drop of production, increase and selectivity in mortality, increase in abundance of older age classes, consequential shift of gender ratio toward females, etc. In Serbia, C&R was introduced into Brown Trout fisheries around 2000, both as a regulatory measure, and as a voluntary act of sportsmanship. Nevertheless, a considerable proportion of fly fishermen are not friendly towards it (Simonović 2015b).

This paper investigates relationships among various stakeholders (e.g., fishermen, fishery managers, trout farm owners, policy makers, administrative managers and conservation scientists) who participate in management of Brown Trout stocks in Serbia. We consider certain questions:

- Is stocking Brown Trout a satisfactory approach to the long-term conservation of native Brown Trout stocks?

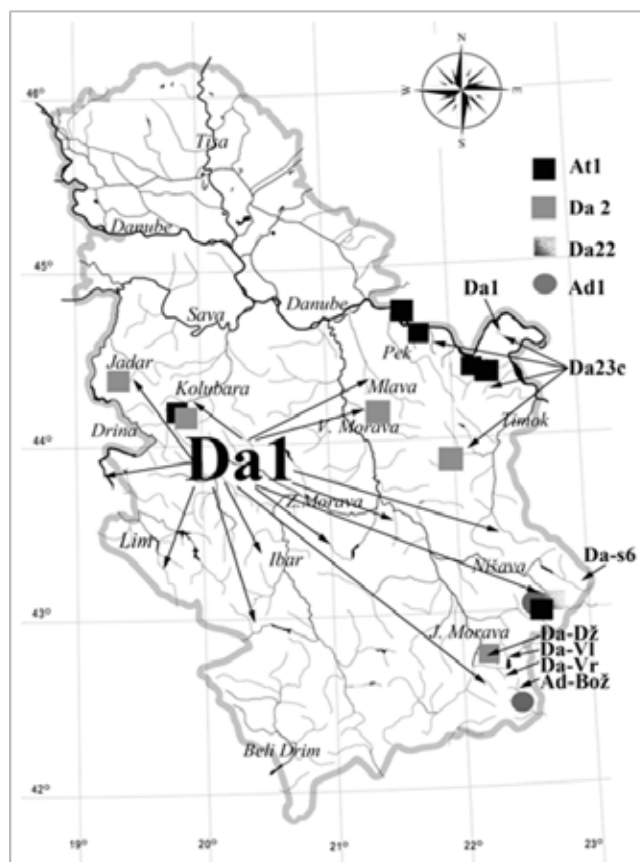


Figure 1. Distribution of native (named and indicated by arrows) and non-native (denoted by symbols) haplotypes in Brown Trout stocks of Serbia (both Ad-Bož and Ad1 haplotype belong to *Salmo macedonicus*).

- Do fishermen insist that Brown Trout fisheries must be stocked for good fishing?
- Do Brown Trout farmers push forward their interest of producing the stocking material?
- Do Brown Trout fishery managers find stocking the easiest way for good fishing?

Materials and Methods

In Serbia, all waters are public and fish as a natural resource are state-owned; only particular fish ponds and fish farms that allow a daily-license based recreational fishing are on private property. Commercial fishing occurs on the few large Pannonian rivers (e.g., the Danube, Sava and Tisa) only. Recreational fishing is allowed everywhere, except in the most strictly protected natural areas. Fisheries are given by public competition to concessionaires for a 10-year-period of management. Concessionaires are

required to prepare management plans for the fisheries and to manage them sustainably, with an emphasis on conservation of the most valuable fish stocks. They sell fishing licenses, of the price determined each year by state fisheries authority. Anglers can choose between buying daily, short-term, or annual fishing licenses. All daily and short-term licenses, as well as the annual licenses issued in the natural protected areas are valid only in the fisheries where they are issued. Whereas, the common annual fishing license issued for recreational fishing in fisheries out of the protected areas are valid in all fisheries throughout Serbia and anglers must buy them in their place of residence, only. For that reason, the vast majority of income for fisheries managers comes from the annual licenses. From that money, fishery managers pay the state tax and cover all costs of management, including the preparation of management plans (long-term and annual ones) and a monitoring of fish stocks. From the tax, state fisheries authorities finance particular projects and activities in concern of fishery management, e.g., strengthening of the capacity of fishery manager, participation in buying of the expensive equipment and in conservation of threatened fish species attractive for fishing, e.g., Huchen *Hucho hucho*.

Brown Trout streams were sampled either by single-pass point-sample electrofishing (Persat and Copp 1989), or fly-fishing, to obtain fin clips for genetic analysis. Brown Trout were measured for their standard length and mass, or photographed, and released. Their length- and weight-frequency distributions were used for ageing, and participation of particular age classes served for calculation of biomass, annual survival rate and annual natural production (Ricker 1975). The potential production of fisheries was estimated after Huet (1975). Brown Trout stock of the River Đetinja was evaluated from images of fly fishing catches of Brown Trout by their von Bertalanffy's (1957) growth parameters and calculation of the L_{opt} , the optimal length that gives maximal yield (Beverton 1992).

Native character of Brown Trout stocks was examined by analyzing their mtDNA haplotypes, using the methodology for DNA extraction and D-loop amplification described in Tošić et al. (2014). The amplified D-loop region of the mtDNA was preliminarily investigated by Restriction Fragment Length Polymorphism (RFLP) analysis using *AluI*

and *SatI* endonucleases and visualization on gel, as explained in Simonović et al. (2015b). Amplified D-loop of each fish in the sample was sequenced by MacroGen Europe Inc.

Nativeness of Brown Trout stocks was used as a measure of aboriginality, and wildness as measure of their self-sustainability without any stocking in the fishery over the long term.

Results

Policy and Legislation in Management and Conservation

As for the policy of management and conservation of Brown Trout stocks, lack of control of the stocking strains and decision to stock all Brown Trout fisheries from the one large hatchery are recently key determinants of the strategy for trout fishery management sought in Serbia.

Brown Trout fishing in Serbia is allowed using only the artificial terminal tackle, principally, flies and lures. The bag limit for Brown Trout is three per day, and size limit is 25 cm SL. Regulations can be additionally restricted by incorporation of C&R into the Management Plan, with the justification as either improving the fishery, increasing protection of the population, or both. The Act on Protection and Sustainable Use of Fish Stocks (Anonymous 2014) states that stocking must be accomplished using the stocking material of only native species, produced in the licensed hatcheries, and declared healthy by authorized veterinary service. In addition, the Code Book on Stocking (Anonymous 2015) considers the stocking material as follows: fertilized roe, fry, brood fish, translocated wild fish, and native fish imported from abroad. It states that stocking material must be healthy, in a good condition, of the even age structure, and in a pure, single species composition. Only for reintroductions, the stocking material must be native such that the brood fish originate from the drainage area of concern.

Stocking as a Threat to Native Brown Trout Stocks

There is no reliable evidence indicating that the Brown Trout Da2 haplotype is native to Serbia. Their uneven distribution only in the productive and the most attractive Brown Trout streams implicates stocking as the main vector of their dispersal. There

is no stream either in the River Danube drainage area of Serbia, or in the Balkans where Brown Trout of Da2 mtDNA is exclusive. Only in the Rivers Gacka and Vrijeka that are situated in the Adriatic Sea slope, Brown Trout of that Da haplotype are exclusive. The native range of Brown Trout of the Da2 haplotype is in streams that confluent with the German (Bernathez 2001) and Austrian upper Danube River (Weiss et al. 2001), where their domestication most likely took place in the late 18th or early 19th century. Kohout et al. (2012) stated that very extensive stockings were initiated in the 19th century during the Austrian-Hungarian Empire in Central Europe, e.g., from the hatchery in Salzburg (Austria) to streams in the North Sea basin. Gridelli (1936) and Razpet et al. (2007) analyzed stocking of streams in Slovenia and Bosnia-Herzegovina, and Mrdak (2011) reported occurrence of Brown Trout of the Da2 haplotype in the most attractive Brown Trout streams in Montenegro.

Data on stocking Brown Trout streams in Serbia are scarce. Stocking started in the second half of the 20th century. Brown Trout of the Da2 haplotype were detected in streams considered attractive for fishing (e.g., in Rivers Gradac, Resava, in headwaters of the River Crni Timok and in the Zmajevac Creek), but also in small streams (e.g., River Džepška) (Marić et al. 2006; Kohout et al. 2013; Tošić et al. 2014). The most recent detection of the ill-advised stocking was the finding of Brown Trout of the Da22 haplotype in the River Jerma (Simonović et al. 2015). Da22 is a strain known only in Brown Trout from Austria and Western Bosnia (Duftner et al. 2003; Škraba et al. 2017). Finally, Brown Trout of the At1 haplotype were introduced first by stocking into the River Gradac in 2001. Afterwards, Brown Trout of the same haplotype were detected throughout Serbia (Marić et al. 2006, 2012; Simonović et al. 2015; Tošić et al. 2016).

Conservational Effects of C&R and Stocking as Management Tools

Before 2000, no Brown Trout fisheries were C&R. The first fishery management plan implementing C&R was that for the River Gradac. Applying only C&R for amenity and conservational reasons revealed extraordinary effects on three Brown Trout fisheries in Serbia. The River Gradac is a freestone headwater in the natural protected area whose wild Brown Trout stock consists of the native Da1 and introduced Da2 and At haplotypes. It was subjected to great fishing

pressure. Without any stocking, only by C&R, the biomass of Brown Trout raised up to 227.7 ± 9.45 kg·ha⁻¹ (Simonović et al. 2014b) in 2003. The River Detinja is a freestone tailwater previously devoid of Brown Trout, where stocked fish became naturalized in a decade, with only protection by C&R. The value of L_{opt} in Brown Trout there increased from 23.1 cm SL in 2012 to 31.8 cm SL in 2016, which suggests that C&R raised the attractiveness of the fishery owing to an upward shift in size structure. The River Crni Timok, a spring creek, supports in its headwaters Brown Trout of the unique Da23c haplotype. Before that was learnt, the fishery was heavily stocked with Brown Trout of the nonnative Da2 haplotype until 2003. The total biomass of trout peaked at 37.6 kg·ha⁻¹, and was extensively exploited by fishing down to 15.9 kg·ha⁻¹ in 2008 (Simonović et al. 2003, 2011). With only the C&R applied, the frequency of Brown Trout catches remarkably increased in last 5 years (pers. observation).

In two Brown Trout freestone tailwater fisheries, Rivers Rasina and Lomnica, a combination of C&R and stocking also revealed auspicious effects. These fisheries were tremendously depleted in their tailwater sections down to 1.2 kg·ha⁻¹ and 2.7 kg·ha⁻¹ and recovered by stocking and C&R in five consecutive years (2011 – 2015) up to 60.4 kg·ha⁻¹ and 40.9 kg·ha⁻¹, respectively. Headwater sections of both streams that contain the native Da1 Brown Trout have not been stocked. Preliminary genotyping of the stocked Brown Trout yearlings revealed they were also of the native, Da1 haplotype.

On the contrary, some stocks important for Brown Trout conservation were compromised by stocking (Figure 1). For example, the River Jerma (SE Serbia) supports native Brown Trout of the Da1 haplotype and has already been stocked with Macedonian trout *S. macedonicus* (Karaman, 1924; Marić et al. 2006). Recently, in addition to them, Brown Trout of Atlantic At1 (Figure 2) and Danubian Da22 haplotypes were found there. In the River Džepška (Southern Serbia), in total four haplotypes were reported (Kohout et al. 2013), compared to only the exclusive Da*Dž haplotype that Marić et al. (2006) discovered.

Certain Brown Trout streams (e.g., Zmajevac Creek in Western Serbia and River Radovanjska in Eastern Serbia) were even overstocked, which led to stunted growth in dense Brown Trout populations (unpublished). Others were pushed beyond their

limits by stocking with oversized brood fish, e.g., the River Gradac in Western (Simonović et al. 2014b), and River Moravica in Eastern Serbia (unpublished). Stocking in 2007 in the hitherto C&R fishery River Gradac decreased Brown Trout density and annual natural production, changed age structure and delayed maturation in young adults, in comparison to the stream section where only fry were stocked. That stocking also introduced more alien Atlantic Brown Trout and destroyed the hitherto wild character of the fishery (Simonović et al. 2014b). Implementing of the C&R had a meager effect there.

Discussion

Policy and Legislation in Brown Trout Conservation

The current attitude towards the conservation of native and wild Brown Trout stocks in Serbia looks opposite to the contemporary principles of management (White 1989; Laikre et al. 1999; Mitro 2004). There are important ecological, scientific, economic, cultural, and moral/spiritual reasons (Bosse, 2004) why the conservation and restoration of native Brown Trout stocks should be undertaken, especially the ones that hold exclusive and specific strains and

those in natural protected areas. Intended stocking from the single, central hatchery would continue the stocking practice that hitherto resulted with deterioration of native Brown Trout stocks. Instead, Simonović et al. (2015) proposed the setting of the state-authorized small-sized hatchery units based on the local Brown Trout brood stocks wherever that is justified. These hatchery units should be operated by the Brown Trout fishery manager, whilst subsidized and controlled by state fishery authorities. Hatchery units should be of the closed type, without any introduction of Brown Trout from outside the range of native strains. Their brood stocks should be genotyped and operated in a small range, to provide only maintenance stocking and to be activated for stocking the domicile streams only. Efforts in spawning and rearing of Brown Trout of the exclusive Da23c haplotype (unpublished), undertaken by enthusiastic fishery manager and scientific collaborators, and supported by fishery authorities, revealed that it is realistic to make an advance in getting roe and milt from wild brood fish, fertilize, incubate, and rear fry to the parr stage, and even to fingerlings. But the successful long-term operation without the system and auspices set by fishery state authorities is not realistic. All conservational activities should be a matter of state policy issued by the legislature, supported by science and other stakeholders.

The Act on Animal Welfare (Anonymous 2009) states that each individual of breeding stock at farms must be tagged and registered by an authorized institution. Investigation into the brood stock of the one of two licensed Brown Trout hatcheries in 2008 revealed a brood fish of Atlantic lineage of both sexes in addition to those of the Da lineage, that have been already tagged and recorded in the Official State Registry. Genotyping of stocking material is not currently demanded by legal regulation of the fishery in Serbia (Anonymous 2014). Genotyping of Brown Trout stocks is still only a matter of science, and not of conservational activities. The matching of the stocking material to the recipient Brown Trout stock should be clearly defined in the fishery legislation.

Brown Trout Conservation, C&R, Stocking and Acting of Stakeholders

If fisheries would be run only as C&R, that might take some extra time for recovery of native or wild Brown Trout stocks on their own, but with no risk



Figure 2. RFLP analysis of Brown Trout from the River Jerma. *SatI* cuts D-loop at one place and gives two fragments (390 and 690 bp) in At Brown Trout, whereas in Da and Ad Brown Trout it does not cut D-loop, which is whole (1080 bp) visualized on the gel. *AluI* cuts D-loop at three places producing four segments (563, 464, 37, and 4 bp) that show two bands in Ad Brown Trout (notably, the last two fragments are too small for visualization on gel). In Da and At Brown Trout, *AluI* cuts D-loop at four places, producing five segments (of lengths 464, 311, 252, 37, and 4 bp) that display as three bands.

of ‘conservational damage’. Nevertheless, inasmuch C&R has good conservational effects on native and wild Brown Trout stocks, it is questioned by various stakeholders. About one-third of fly fishermen in Serbia are not satisfied with a mandatory, total C&R (Simonović et al. 2014a). They are used to bag- and size-limits (i.e., a conditional C&R) and a majority of them strongly demand stocking, even in the streams where the total C&R is in effect. They truly believe that stocking is the most efficient way to maintain a Brown Trout fishery in a good shape and neglect the purpose, role and effects of the C&R regime of fishing. That reveals their unfamiliarity with the conservational dependence of Brown Trout stock management, which makes it difficult to negotiate with them. Hatchery managers endorse stocking to enhance sport fisheries and fishery managers support stocking as long as that improves incomes and profit. That affects especially the attractive Brown Trout fisheries, leads to their commercialization and destroys the native character of their stocks. Even though fly fishermen may express support for C&R, they clearly are opposed to C&R regulations that may require extended periods of time to see marked improvements in the fishery. When C&R extends limitations to their fishing opportunities, they withdraw their support of conservational activities (Simonović et al. 2014a). Fisheries with oversized (old hatchery brood fish) and overcrowded Brown Trout stocks reflect responses of fishery managers from, and their implicit settling to fly fishermen who refuse to stick to the C&R regime. Two fisheries where C&R was issued were nevertheless stocked every year with brood fish of the trophy size, excessive for streams of the small size and productivity. Although poaching was a likely source of Brown Trout mortality, it is also likely that in densely stocked tailwater fisheries Brown Trout moved downstream looking for vacant room and available food. That perpetually decreased the attractiveness of fishing and forced the fishery manager to stock every year.

Nevertheless, findings about the native character of Brown Trout stocks were implemented in particular fisheries management plans and justified (e.g., by risk the stocking poses to headwaters with pure, native stocks, or to those with exclusive haplotypes), to replace stocking with the C&R.

Legal mandates of stocking with only native fish species (Anonymous 2014) drops the likelihood of alien trout species introduction into the streams of

Serbia. However, even the Code Book on Stocking (Anonymous 2015) does not oblige the use of native Brown Trout stocking material, except for reintroductions. Considering the isolation and local-scale differentiation of Brown Trout populations, it is difficult to provide the appropriate stocking material for reintroduction. Alternatively, the repeated translocation of very limited number of the similar native Brown Trout fish from the adjacent streams would be satisfactory to initialize re-founding of the population and of the gene pool as similar as possible to the original one.

A common driving force to stocking is fishermen’s desire to catch a trophy-sized Brown Trout, regardless of stream productivity. Fishery managers do not hesitate to pay for stocking material if that maintains or increases license sales from satisfied fishermen, and trout farmers readily produce stocking material for them. The income is an impetus that promotes stocking of Brown Trout fisheries. That can be turned in favor of native or wild Brown Trout stocks by providing better incomes for fishery managers who run native or wild Brown Trout fisheries attractive for fishing. That would promote the maintenance of Brown Trout native and wild. Success in promoting that would be partly a matter of market, but also of the support from the state fishery authorities, either by opportunity given to fishery managers to issue more expensive fishing licenses, or by subsidizing them. In every fishery system there is an opportunity for subsidies from the portion of license fees that fishery managers pay into the state budget. That can provide a sustainability of conservational efforts over a long time period.

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Session 2:

Threats and Management of Stream Habitat: A Look Into the Future





The Southern Appalachian Brook Trout Management Conundrum: What Should Restoration Look Like in the 21st Century?

By Matt A. Kulp¹, Shawna Mitchell², Dave Kazyak³, Bernard Kuhajda², Jason Henegar⁴, Casey Weathers⁵, Anna George², Josh Ennen², Tim King³

¹Great Smoky Mountains National Park, Gatlinburg, TN, Matt_Kulp@nps.gov

²Tennessee Aquarium Conservation Institute, Chattanooga, TN

³U.S. Geological Survey, Leetown Science Center, Kearneysville, WV

⁴Tennessee Wildlife Resources Agency, Nashville, TN

⁵Penn State University, State College, PA

Abstract—Brook Trout *Salvelinus fontinalis* in the southern Appalachian portion of their range have been isolated in remote headwater systems for millennia. Recent genetic investigations indicate extremely low allelic diversity, heterozygosity and effective population sizes in many streams. In populations restored using multiple source stocks, limited introgression has been observed despite source stocks being collected from streams within the same subwatershed. It remains unclear if pre- and/or post-reproductive isolating mechanisms are restricting effective gene flow among source stocks in restored streams. Objectives of this study were to: 1) identify environmental variables contributing to assortative mating, and 2) use common garden crossings to determine if wild type brood stock crossings resulted in physiologically viable offspring. We observed markedly different fertilization success rates within-population (66.7%) and between-population (91.7%) from the 42 crosses (N=18 control, N=24 treatment). Moreover, we observed significant ($P < 0.05$) differences between within-population and between-population groups in each of our linear mixed effects global models for each trial stage of development (i.e., fertilization rate, eyed egg rate, and hatch rates). Tukey's HSD comparisons revealed only one significantly ($P < 0.003$) different fertilization rate among the forty five pairwise comparisons in each of our three stages of trials. In addition, we observed differential peaks of gamete production within and among source stream brood stock, despite common garden conditions, that appeared to have limited fertilization success rates between interstream and control groups. Despite differential peak gamete timing, intrastream crosses performed equally, and, in some instances, better than those between control groups. Our results suggest differential responses to shared environmental conditions (i.e., temperature and/or photoperiod) may contribute to mismatched spawning phenology (i.e., gamete production timing) among restoration founder stocks leading to introgression (i.e., genetic admixture). The application of contemporary genetic techniques could help determine if these possible local adaptations are genetically fixed or may break down over time in restored populations with mixed source stocks. These findings demonstrate the need to apply contemporary conservation genetics tools to future wild trout restoration projects using translocated source stock towards the goal of "genetically-robust", naturally reproducing populations with the ability to cope with current and future perturbations.

Introduction

The protection and preservation of native species is a primary management goal of the National Park Service (NPS) dating back to the Organic Act of 1916 (16 U.S.C. §1). NPS policy is unique among land management agencies in that the NPS is mandated to protect and preserve "naturally functioning ecosystems", which includes the removal of nonnative species (NPS Management Policies 2006). Prior to the establishment of Great Smoky Mountains National Park (GRSM) in 1934, many watersheds

were extensively logged (Pyle 1988) and streams across the park were stocked with nonnative Rainbow Trout *Oncorhynchus mykiss* (King 1938; Etnier and Starnes 1993). These actions resulted in Brook Trout *Salvelinus fontinalis* being extirpated from 75% of their native range within GRSM by the 1980s (Moore et al. 1986). Given the negative impacts on native Brook Trout, in the late 1950s, the U.S. Fish and Wildlife Service and NPS began taking steps to remove nonnative Rainbow Trout from park streams in order to restore Brook Trout back to portions

of their former range (Lennon and Parker 1959). Rainbow Trout removal techniques have included angling (Larson et al. 1986; Moore et al. 1986), annual removals using backpack electrofishing gear (Moore et al. 1986; West et al. 1990) and the use of piscicides such as rotenone, cresol and Fintrol® (a.k.a. antimycin-A) (Lennon and Parker 1959; Moore et al. 2005; Vinson et al. 2010). To date, GRSM has restored 44.2 km of 11 streams by removing nonnative Rainbow Trout and relocating southern Appalachian Brook Trout from headwater areas of the restored stream or translocating Brook Trout from multiple sources stocks of other streams within the major subwatershed (Kanno et al. 2016).

Most remaining Southern Appalachian Brook Trout populations within GRSM are relegated to highly fragmented, headwater stream segments with little to no metapopulation structure (Weathers et al. *In Review*). These highly fragmented populations exhibit limited genetic exchange (Kazyak et al. 2016) and poor allelic diversity (Richards et al. 2008). Demographically, most GRSM allopatric Brook Trout stream populations support low to moderate densities (i.e., 4-32 fish/100m²) (Kanno et al. 2017) in comparison to allopatric Brook Trout populations from more productive streams across their range (10-70 fish/100m²) (Kocovsky and Carline 2006; Petty et al. 2014). Despite the low genetic diversity and moderate population abundance of most GRSM Brook Trout populations, local source stocks are still needed for translocation to newly restored stream segments across the park where restoration is feasible.

In 2000, roughly 120 to 150 Brook Trout from each of three source streams [Cosby Creek (C), Greenbrier Creek (GB) and Indian Camp Creek (IC)] were translocated into Leconte Creek after nonnative Rainbow Trout were removed via backpack electrofishing. In 2008, attempts were made to describe the genetic composition of the Leconte Creek population in order to determine whether each population was equally represented or if one of three parental populations (i.e., Cosby Creek, Greenbrier Creek and Indian Camp Creek) may be better adapted to the conditions of Leconte Creek (Richards et al. 2008). Parentage analyses 7 and 11 years post stocking indicated that 76% and 82% of the sampled LC fish were produced from matings between parents from the same stream of origin (Richards et al. 2008; *Tim King personal observation*). These results

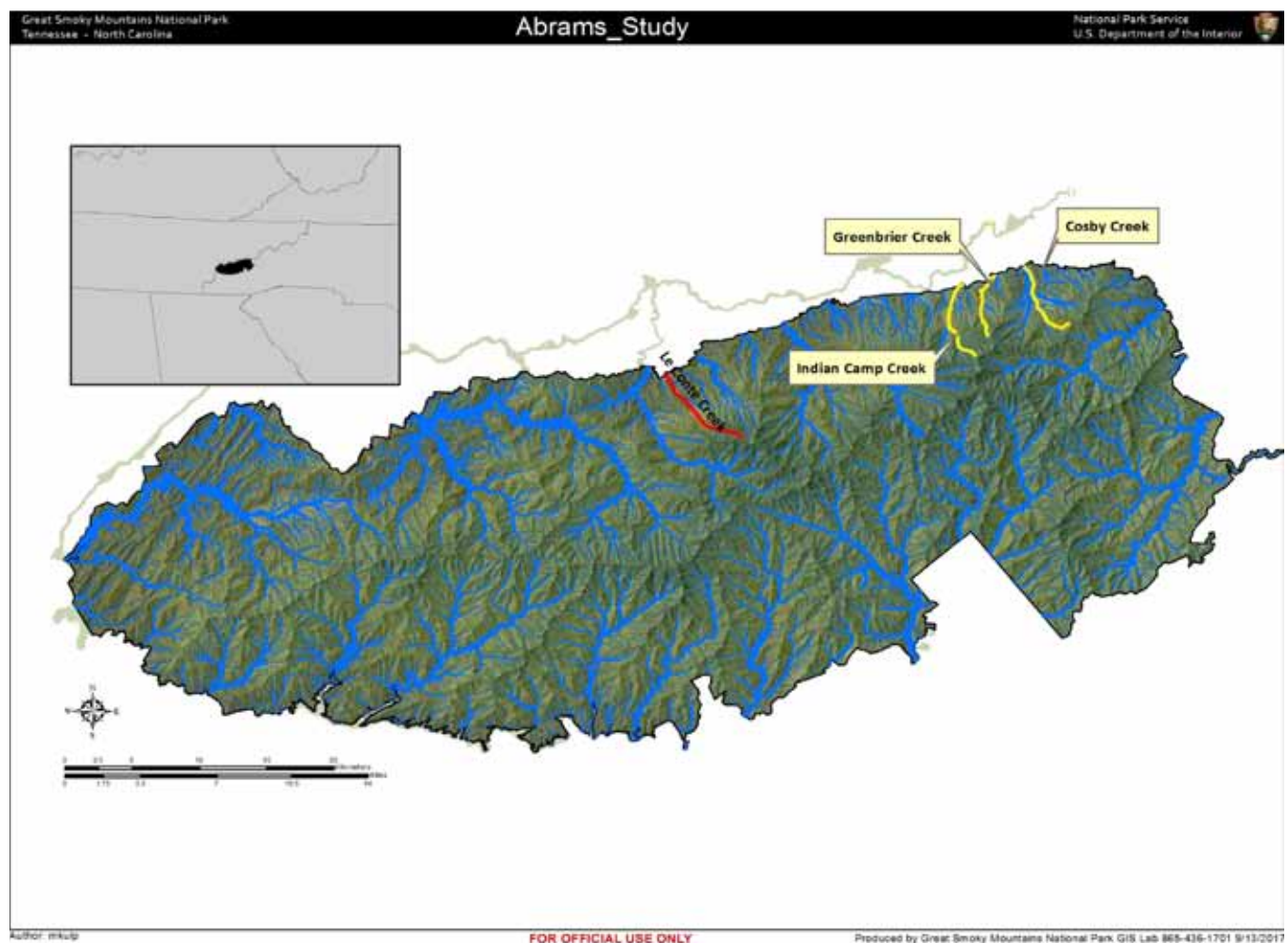
suggest the presence of either a pre-reproductive isolating mechanism(s) (i.e., environmental cue for spawning, positive assortative mating) or the presence of some type of post-reproductive isolating mechanism(s) (i.e., loss of locally-adapted gene complexes, outbreeding depression).

Despite compelling genetics data to suggest source stock populations from across the park are genetically distinct from one another (Richards et al. 2008; Weathers, Personal Communication), the need remains to restore populations using multiple source stocks as population densities are typically too low to collect hundreds of individuals from a single source for population translocation. In doing so, managers face the conundrum as noted by Houde et al. (2011) of either limiting restoration source stocks to a single small, fragmented population, which risks continued inbreeding or collecting and mixing multiple source stocks, which risks outbreeding depression in subsequent generations. In weighing the conundrum outlined by Houde et al. (2011), GRSM managers were unclear if the observed assortative mating findings (Richards et al. 2008) were the result of pre- or post-reproductive isolating mechanisms. Therefore, further information was needed to enhance the effectiveness of future restoration efforts. In order to clarify which of these issues may have contributed to limited admixture, study objectives were to: (1) identify environmental variables contributing to assortative mating, and (2) use common garden crossings to determine if wild type brood stock crossings resulted in physiologically viable offspring.

Study Area

Great Smoky Mountains National Park was established in 1934 and currently encompasses 211,039.52 hectares of southern Appalachian hardwood forests in eastern Tennessee and western North Carolina (Figure 1). The park includes 45 major watersheds (>0.5 km²), all of which eventually flow into the Tennessee River. The more than 4,640 km of streams within GRSM range from first to sixth order and include both cold and cool-water stream environments (Kanno et al. 2017). Stream gradients typically range from 2% to 20% throughout the park, and daily stream discharges range from 0.03 to 23 m³/s. Water temperatures typically range from 0.8 °C to 18.8 °C; however, summer water temperatures occasionally reach critical limits for salmonids (>25.8 °C) during

Figure 1. Map of Great Smoky Mountains National Park including newly restored Leconte Creek and locations of Brook Trout source stock streams (i.e., Cosby Creek, Greenbrier Creek and Indian Camp Creek).



drought periods in some lower elevation (<450 m) stream segments (fifth to sixth order). Dominant substrate includes small boulder, large boulder, and cobble, with little or no instream vegetation. Precambrian sandstone dominates the underlying geology of GRSM, which includes several areas of exposed and unexposed acidic Anakeesta Formation and a few small windows through Ordovician limestone. Given the underlying geology, the poor buffering capacity of these formations, and some of the highest acid deposition rates in North America (Robinson et al. 2008; Fakhraei et al. 2016), the pH of GRSM streams typically ranges from below 5.0 to 6.5 throughout much of the park; stream conductivity is less than 30 $\mu\text{S}/\text{cm}$ (Robinson et al. 2008).

Cosby Creek, Greenbrier Creek and Indian Camp Creek are each third order tributaries of the Pigeon River, a tributary to the Little Tennessee River system (Figure 1; Table 1). Brook Trout in each stream were previously genotyped using microsatellites and unique markers established by King *et al.* (2012) to ensure source stocks had no previous hatchery introgression. These streams were selected as they provided modest population abundance (biomass range 16-68 kg/ha), modest effective population sizes (mean = 23.3) and reasonable access.

Methods

Attempts were made to collect roughly 20 male and 10 female Brook Trout from each of the three

Table 1. Physical and chemical characteristics of the three source streams from which southern Appalachian Brook Trout were translocated during Brook Trout restoration efforts within Leconte Creek, Great Smoky Mountains National Park. Values in parentheses are standard errors (SE). Aluminum and calcium values were not available from Greenbrier Creek or Leconte Creek.

Stream Name	Order	Elevation (m)	Mean Width (m)	Mean Gradient (%)	Mean Discharge (m ³ /sec)	Mean Conductivity (μS/cm)	Mean pH	Mean ANC (μeq/L)	Aluminum (mg/L)	Calcium (μeq/L)
Cosby Creek	3	804	4.9 (0.6)	10.4 (0.4)	0.102 (0.01)	17.2 (0.26)	6.20 (0.05)	33.2 (1.26)	0.06 (0.01)	66.9 (1.2)
Greenbrier Creek	3	742	4.5 (0.2)	8.5 (0.3)	0.052 (0.00)	20.2	6.74	48.4	N/A	N/A
Indian Camp Creek	3	917	7.6 (0.8)	8.0 (0.8)	0.161 (0.01)	16.2 (0.26)	6.01 (0.05)	20.2 (1.27)	0.06 (0.01)	64.5 (4.7)
Leconte Creek	2	673	4.6 (0.2)	8.4 (0.5)	0.084 (0.02)	15.9 (1.10)	6.32 (0.05)	36.37 (3.58)	N/A	N/A

source streams using backpack electrofishing gear (Kanno et al. 2017) in the spring of 2015. Actual 2015 collections included: Cosby Creek (12 females, 11 males), Greenbrier Creek (8 females, 28 males) and Indian Camp Creek (11 females, 22 males). Once collected, fish were placed in a bucket with 11.4 L of water and anaesthetized using 0.5ml of clove oil (95% ethanol:5% clove oil) and 12 mm 134.2 kHz Biomark PIT tags were injected subcutaneously into the muscle above the lateral line, anterior to the dorsal fin so that origin and sex could be identified. Once tagged and recovered in fresh water, Brook Trout were placed in backpack transporters, carried out to an awaiting hatchery truck and transported to the Tennessee Aquarium Conservation Institute (TNACI). Due to first year holding mortality, and an attempt to meet the minimum number of individuals per sex for crosses, additional brood stock collections were made in 2016 (C 5 female/5 male; GB 5 female/0 male; IC 10 female/0 male). Once at the aquarium, fish were placed into two 530 L holding tanks and fed daily rations of minced frozen krill.

In order to replicate natural conditions, broodstock holding tanks were gradually brought from 10 °C (May) to 12.8 °C (August) in 2015. In mid-August they were decreased from 12.8 °C to 11.7 °C where they were held at that temperature until early October when they were decreased to 11.1 °C and held there until the end of January 2016. In 2016, common garden temperature conditions were changed to better mimic stream temperature conditions observed in the field during the 2010, 2011 and 2012 spawning seasons. Field spawning and mating pair behavioral patterns were recorded across the fall 2010, 2011

and 2012 spawning seasons to infer when fish would stage and spawn in the common garden setting. From May to August 2016, holding tanks were gradually raised from 11.7 °C to 14.4 °C. On September 1st, the tanks were decreased to 13.3 °C then 6 days later were decreased to 12.2 °C. In early November, the temperature was then decreased another degree to 11.1 °C where it was held through the end of December. Spawning crosses were attempted during the first week of October each year. Prior field stream temperatures (January 2010 through March 2012) were recorded in each source stream and Leconte Creek in order to describe field temperature fluxes and simulate common garden conditions.

Nine female Brook Trout from each source stream were crossed with males from each of nine study groups (i.e., 3 controls, 6 treatments), which included 3 replicates per group, resulting in a total of 27 potential crosses. Attempts were made to produce the 27 crosses (9 controls, 18 treatments) in both 2015 and 2016. Crosses began in October 2015 as water temperatures were lowered to 11.1 °C. Fish were placed into 8 plastic holding bins on 2 large plastic carts (378 L), fitted with an aerator and filled halfway with water from the sump from one of the recirculating systems. Fish were sedated using a 100 ppm MS-222 bath and sodium bicarbonate (2x the grams of MS-222) to neutralize pH. Broodstock were netted out of a holding tank, scanned for their PIT tag with a Biomark 601 handheld PIT tag reader, referenced to a list that had broodstock sorted by sex and stream and then placed into their respective holding bin.

Once all fish were sorted, a randomized spawning list was referenced and the first cross pairing that

had not been attempted was selected. Two to three males were randomly taken from their holding bin with an aquarium dipnet and placed into the MS-222 bath. Once fish lost equilibrium, one male and one female were each randomly taken out of the bath by hand, measured (mm) and weighed (g). A single male or female was then stroked 3 times with its vent directed at a clean spawning bowl. If no gametes were produced, the fish was placed in the “hard” trough. If gametes were produced, the fish PIT tag was scanned and the fish placed into the male holding bin. If eggs were fully developed and easily came out of the female, she was stroked until 3 strokes resulted in no more eggs exiting her vent (Figure 2). After a male and female was placed into the holding bin, remaining males and females in the MS-222 bath were returned to their holding bin to recover. The female was then weighed again on a scale (g) and PIT tag scanned before being placed into the “spent” cart. The male

Figure 2. Picture of Tennessee Aquarium Conservation Institute (TNACI) biologist Shawna Mitchell stripping eggs from a female Southern Appalachian Brook Trout as part of an interagency Brook Trout restoration crossing study.



that contributed to that cross was then moved from the male holding bin to the “spent” cart.

To activate egg fertilization, saline water (32 ppt salt) was poured into the spawning bowl until water covered the eggs. After 5 minutes, the contents of the spawning bowl were poured into a soft mesh aquarium dip net that was immersed in a 19 L bucket filled three fourths capacity with fresh water from one of the recirculating systems. The eggs were gently rinsed to remove excess milt, ovarian fluid, and salt water by very gently moving the net in and out of the water 5 times. Eggs were then gently placed into a 1000ml glass measuring cup that had a 50ppm Ovadine (1 % active iodine) solution in it. The eggs were allowed to water harden and disinfect in the Ovadine solution for 15 min, rinsed to remove excess Ovadine and then transferred into a glass measuring cup filled with 500 ml of water from the one of the recirculating systems. The randomized spawning list was referenced to learn the hatching jar number that was assigned to each specific cross and the eggs poured through a plastic funnel into their respective hatching jar. Water flow was adjusted so eggs gently rolled in the jar and black plastic placed on top of the jar to prevent light from affecting egg development.

Visual inspections were used to determine the number of crosses successfully fertilized and results recorded. As eyed eggs developed from successfully fertilized crosses, they were checked daily and dead eggs were carefully removed and recorded. Eggs that hatched from eyed eggs were gently dumped into a small plastic bin and numbers recorded. Once fry absorbed their yolk sac, they were euthanized with MS-222 and preserved in RNAlater.

Three global linear mixed-effect models (function lme, package nlme, R v 3.3.2) (Pinheiro et al. 2017) were fit to each of the three single trial stage response variables (e.g., percent fertilized, percent eyed, percent hatched) with treatment (fixed effect) and year (random effect). Response variables were arcsine transformed to improve model fit. In instances where our global models revealed a significant difference among treatment groups, we examined which treatment combination(s) contributed to such a statistical result. More specifically, we conducted a pairwise Tukey’s HSD test to identify whether a significant ($P < 0.003$) difference of each treatment group mean existed in R using the package multcomp (Hothorn et al. 2008).

Table 2. Table summarizing the number of successful control and treatment crosses of Brook Trout from three different source stocks. Attempts were made to collect 3 replicates per study group (9 control and 18 treatment groups) and conduct crosses in both 2015 and 2016 for a potential total of 27 crosses per year. Successful crosses were defined as those in which male and female gametes were produced and crossed. Mean percent fertilized, eyed and hatched represents the percentages of eggs meeting that criterion for each group.

	Total Successful Crosses (2015 & 2016)	Number Successfully Fertilized N (%)	Mean Percent Eggs Fertilized per Pair (%)	Number Successfully Eyed N (%)	Mean Percent Eggs Eyed per Pair (%)	Number Successfully Hatched N (%)	Mean Percent Eggs Hatched per Pair (%)
Control	18	12 (66.7%)	85.7%	10 (83.3%)	55.5%	9 (90.0%)	56.1%
Treatment	24	22 (91.7%)	83.2%	18 (81.8%)	37.7%	18 (100.0%)	30.4%
TOTAL	42	34 (80.9%)		28 (82.3%)		27 (96.4%)	

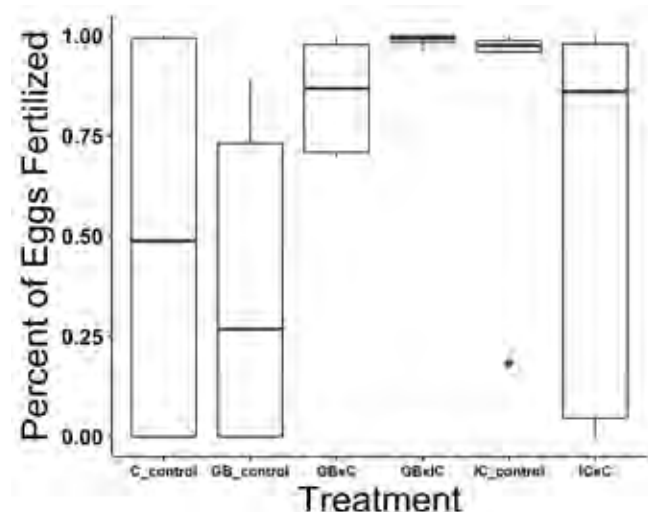
Results

Fourteen of the initial 92 PIT tags (15%) were rejected from the 2015 brood stock. An additional 24 individuals died due to unknown causes (N=11), stress/weight loss (N=7), tag site infection (N=3), predation (N=2) and escape (N=1).

Both control (66.7%) and treatment (91.7%) groups were successfully crossed across all streams in the laboratory setting (Table 2). Percentages of successfully fertilized eggs within pairings were

similar between control (85.7%) and treatment crosses (83.2%). The global model indicated significant differences in fertilization rates ($P < 0.001$; $df = 5$; $\text{ChiSq} = 33.176$) between within-stream (66.7%) and between-stream (91.7%) groups. Similarly, the global eyed egg rate model indicated significant ($P = 0.007$; $df = 5$; $\text{ChiSq} = 15.857$) differences between within-stream (83.3%) and between-stream (81.8%) groups. In addition, the global hatch rate model indicated significant ($P = 0.004$; $df = 5$; $\text{ChiSq} = 17.166$) differences between control and treatment groups. Levene's tests revealed unequal variances among treatments groups.

Figure 3. Box plot of mean percent of Brook Trout eggs fertilized among control and treatment groups for three source stock streams within Great Smoky Mountains National Park. Source stock streams are represented by Cosby Creek (C), Greenbrier Creek (GB) and Indian Camp Creek (IC).



Within each global model, only one of fifteen pairwise fertilization rate comparisons (GBxIC to GBxGB) had statistically significant differences ($P < 0.003$) in means (Figure 3). In all other pairwise comparisons (i.e., 30 of 30), eyed egg (Figure 4) and hatch rate (Figure 5) success means did not significantly differ. Because only 1 of 45 comparisons was significantly different, we concluded that there was no overall difference among within and between-stream groups and chose not to correct for multiple pairwise comparisons (Perneger 1998).

Cooling tank temperatures of 13°C in both 2015 and 2016 initiated gamete production by trout from all three streams, however the period of gamete production varied considerably among years. Water temperatures in 2015 were gradually lowered from 13°C to 11°C over a period of 90 d, whereas temperatures were lowered from 14.4 to 12.2°C in 6 d in 2016 (Figure 6). In 2015, gamete production began at a temperature of 12.8°C with Cosby Creek on 16 August, followed by Indian Camp Creek (1 Sept)

then Greenbrier Creek (19 Oct) (Figure 6). Gamete production continued for adults from each stream at 11.1°C through early January 2016, except for Cosby Creek males, which ceased gamete production in early November 2015. In 2016, gamete production began on 16 August for Greenbrier Creek males and Indian Camp males and females (Figure 6). Cosby Creek males and females began gamete production on 1 Sept when temperature reached 13.3°C. Gamete production for Greenbrier Creek females did not start until 13 Oct when temperature reached 12.2°C. Gamete production for adults from all streams ceased around 19 Oct and did not resume even after temperatures declined to 11.1°C through January 2017.

Conclusions

Both within-stream and between-stream groups were successfully crossed in the laboratory setting. Although there was one statistically significant difference in success among pairwise comparisons, our observations do not suggest that outbreeding depression is limiting interbreeding among the founder stocks. The abundance of successful between-stream crosses at each stage of development suggests that, within Leconte Creek, low genetic admixture

Figure 4. Box plot of mean percent of Brook Trout eggs reaching “eyed” stage among control and treatment groups for three source stock streams within Great Smoky Mountains National Park. Source stock streams are represented by Cosby Creek (C), Greenbrier Creek (GB) and Indian Camp Creek (IC)

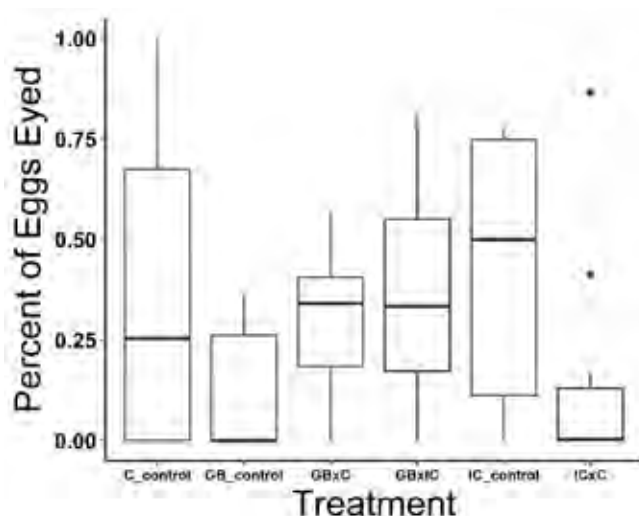
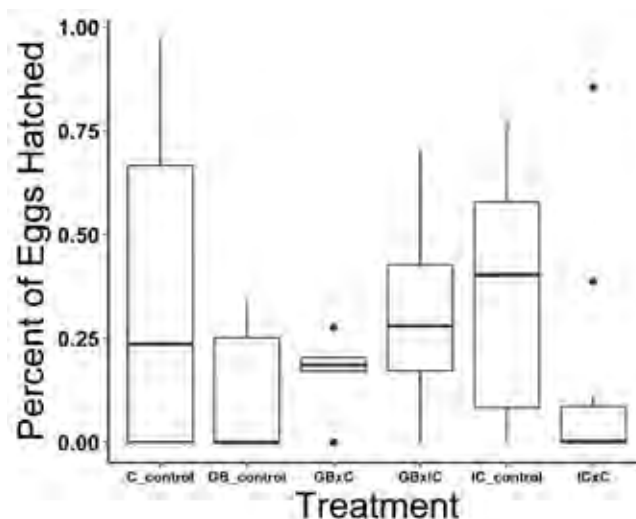


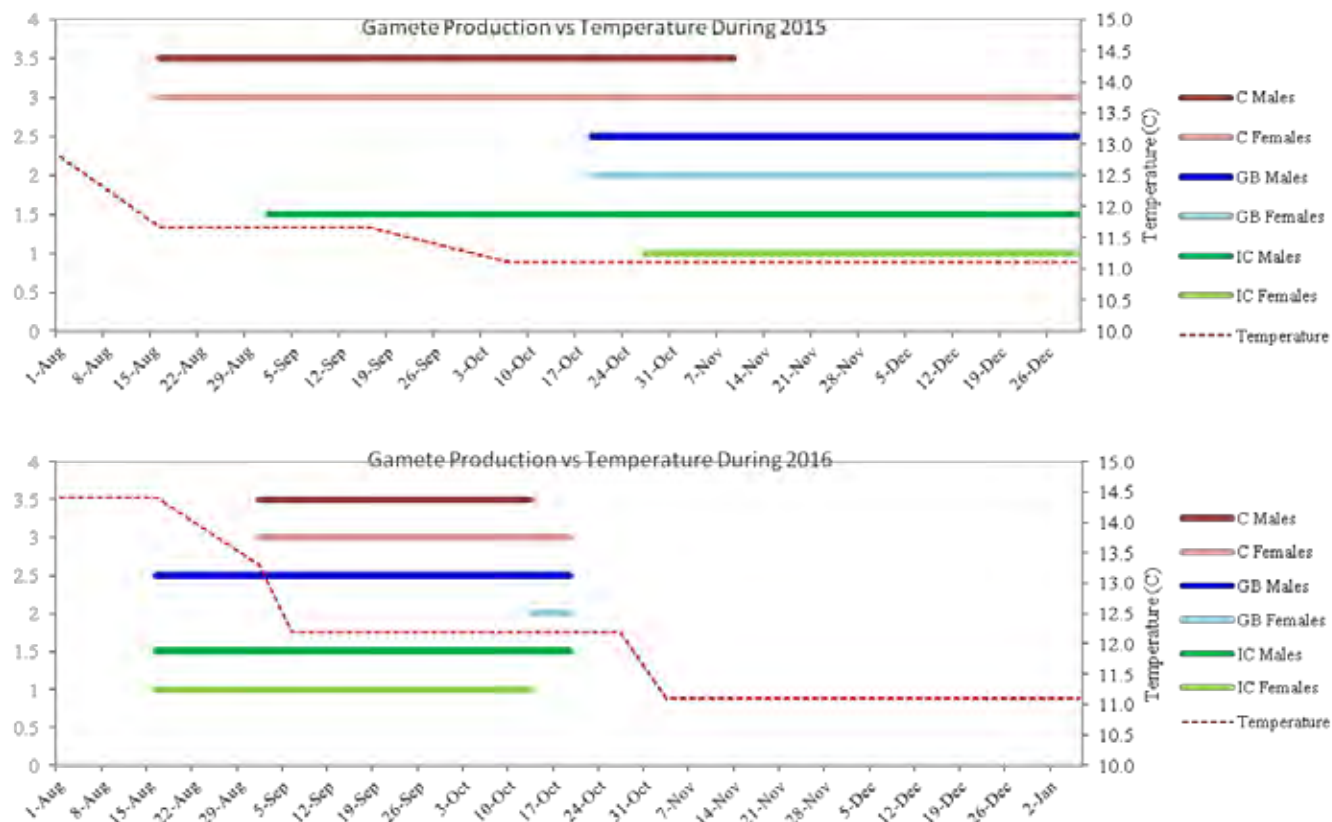
Figure 5. Box plot of mean percent of Brook Trout eggs that reached hatching stage among control and treatment groups for three source stock streams within Great Smoky Mountains National Park. Source stock streams are represented by Cosby Creek (C), Greenbrier Creek (GB) and Indian Camp Creek (IC).



(Richards et al. 2008) is most likely the byproduct of a physiological response to environmental spawning cues (i.e., stream temperature) in Brook Trout.

There are a variety of factors that could have contributed to the limited and differentially timed gamete production observed in 2016. Differences in gamete production may have been the result of poor brood stock fitness due to prolonged captivity (i.e., 5-17 months) of wild fish in the laboratory setting (Blanchet et al. 2008). Wild Southern Appalachian Brook Trout have historically been very difficult to captively rear and spawn in the laboratory setting (Jason Henegar, personal observation). Under or over-maturity of some females coupled with small sample sizes may have also contributed to low and differential 2016 gamete production (Horreo et al. 2008). Long term (>10,000 years) isolation and extensive population fragmentation may have led to local adaptations that are inhibiting local populations from successfully mating, despite being located in close proximity (<16 km) to each other. Field observations of spawning pairs coupled with temperature observations suggest that significant temperature declines during September (i.e., 3-7 °C) trigger

Figure 6. Gamete production of Brook Trout from three different source streams in 2015 and 2016 following the application of two different temperature regimes. Field observations indicate major temperature decreases occur in mid-October and spawning activity ceases in early November. Source stock streams are represented by Cosby Creek (C), Greenbrier Creek (GB) and Indian Camp Creek (IC).



increased gamete production and spawning activity (Weathers and Kulp, personal observation). Under gradual cooling conditions, however we observed that gamete production lasted >120 d. Alternatively, when field temperatures were simulated in 2016, gamete production was initiated earlier and production was much shorter (i.e., ≤60 d). The 2016 laboratory trials reflect typical fall temperature drops (i.e., 3-7 °C) associated with fall rain events. Laboratory notes indicate gamete production was consistent with a normal distribution where a few individuals started producing gametes, production peaked and then tailed off until it ended. Given these patterns, and the slight timing differences noted in 2016 gamete production, it is understandable why it was difficult to fulfill crosses for all study groups. The slightly differential gamete

production observed in our experiment may also be occurring in Leconte Creek. Our findings suggest local adaptation may also be occurring, which is particularly strong in salmonids (Hendry et al. 2003). Locally-adapted responses to thermal cues leading to mismatched reproductive phenology would explain the assortative mating observed over several generations. Other authors (Allendorf and Waples 1996; Fraser et al. 2011) have noted the need to adaptively ‘match’ populations to their new environment in order to increase the potential for restoration success. Although the physical and chemical habitat of Leconte Creek appears to ‘match’ the source streams very closely, there may be subtle molecular differences among populations that remain undetected that are contributing to the assortative mating.

Although crosses of locally adapted populations in newly restored stream segments may initially result in reduced fitness or survival, interbreeding of source stocks may be acceptable in the long term as outbreeding depression may not continue in subsequent generations (Houde et al. 2011). Hybridization of isolated source stocks may result in heterosis, or hybrid vigor that may be retained or lost in subsequent generations (Allendorf et al. 2013), whose effects however are most problematic and poorly understood in population genetics (Allendorf 2017).

Translocation of species in order to restore and/or enhance populations is a relatively longstanding science having been around for roughly 200 years (Hendry et al. 2003; Ewen et al. 2012). Vertebrate translocation projects aimed at restoring populations have increased 342% since the early 1900s, however only 25-30% of these projects are successful at restoring the target species (Houde et al. 2015). Despite the long history of species restoration through reintroductions, there remains a scarcity of data regarding reintroduction outcomes, gradients of genetic similarity, environmental similarity and heritable genetic variation (Houde et al. 2015). Given the advent of recent genetic tools and techniques that now allow managers to sequence entire genomes (Allendorf 2017), we suggest that managers embrace these tools to enhance and improve restoration outcomes.

As restoration efforts in Leconte Creek have shown, translocation of fish and subsequent production of successive generations do not capture the complete story of restoration “success.” Species restoration success should not be measured simply upon population demographics, rather should strive for a “genetically-robust”, naturally reproducing population with the ability to cope with current and future perturbations (Pavlova et al. 2017). Houde et al. (2015) notes there is also a need to determine the role of adaptive capacity in translocation outcome and whether populations with high heritable genetic variation are more likely to re-establish a population in the restored stream section than populations with low heritable genetic variation. Based upon these observations, we suggest future translocations of fish to restore populations include: (1) attempts to capture as much of the genetic variance (i.e., allelic richness, effective population size, expected heterozygosity, etc.) of source stocks in the newly founded population

as possible, (2) incorporating genetic monitoring of newly restored populations in order to assess the level of genetic variation captured in the newly formed population, determine if inbreeding depression is increasing or if outbreeding depression is limiting genetic fitness over time and (3) consideration of continued translocations in order to emulate the effect of historical gene flow thereby improving population persistence, if population connectivity is a management concern. These efforts should help maintain or increase effective population size, decrease demographic and genetic stochasticity and maximize species adaptive potential (Pavlova et al. 2017). Future studies should also investigate if local adaptations, such as those that may be occurring among Leconte Creek source stocks, are fixed or if these pre-zygotic barriers are naturally ameliorated over time as source stocks locally adapt to restored stream segments. These principals will help guide future reintroduction restoration efforts for all species across their range.

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Efficacy of Brook Trout Introductions into Virginia Streams Impacted by a Catastrophic Climatological Event

Steve Owens^{1*}, John Odenkirk², and Mike Isel²

¹Virginia Department of Game and Inland Fisheries, 1796 Highway Sixteen, Marion, Virginia 24354,

*Correspondence author: steve.owens@dgif.virginia.gov

²Virginia Department of Game and Inland Fisheries, 1320 Belman Road, Fredericksburg, Virginia 22401

Abstract—A catastrophic weather event during June 1995 produced as much as 60 cm of rain over 5 d in Madison County, Virginia, resulting in severe flooding and massive debris flows. As a result, wild Brook Trout *Salvelinus fontinalis* were extirpated due to significant habitat alteration in Garth Run and Kinsey Run. Despite habitat improvements 10 years post storm, Brook Trout failed to recolonize study waters from the nearby Rapidan River. Approximately 100 wild Brook Trout of various sizes (mean TL=119 mm; SD=36) were collected by biologists with the Virginia Department of Game and Inland Fisheries (VDGIF) from a neighboring watershed and introduced into each stream during September 2008. Successful reproduction was documented in both streams the following June. Brook Trout recruitment variability was not significantly different between years ($P = 0.001$), but was significantly different between streams ($P = 0.001$) and populations continued to increase 8 years post reintroduction. Adult Brook Trout numbers were variable as initial electrofishing catch rates declined before populations began expanding 7 years post reintroduction. These data exemplify the effectiveness of utilizing small stockings of wild fish into suitable headwater streams as a restoration tool.

Introduction

Brook Trout *Salvelinus fontinalis* are the only salmonid native to Virginia and much of the eastern United States. Generally, Brook Trout populations in Virginia are restricted to headwater streams with moderate to high elevations (>305 m), low siltation, and good water quality associated with the Appalachian Mountains in the western part of the state (Jenkins and Burkhead 1993). Human activities over the past century including logging and development have resulted in the degradation or loss of vital Brook Trout habitat (Wesner et al. 2011). Additionally, many streams in the central Appalachian region approach the upper thermal limit of Brook Trout during summer and trout rely on canopy cover and complex instream habitat (Ries and Perry 1995). In late June 1995, a 5-d rainfall event in the Graves Mill area of Madison County, Virginia produced as much as 60 cm of rain. Severe flooding, debris slides (Figure 1), and drastic changes in stream habitat of Garth Run and Kinsey Run occurred leading to extirpation of resident wild Brook Trout populations (Roghair et al. 2002; Isel 2011). Quantitative and qualitative electrofishing surveys of both streams by the Virginia Department of Game and Inland Fisheries (VDGIF) in 2000 and 2007

resulted in the collection of no Brook Trout (VDGIF, unpublished data; Isel 2011).

Water quality and trout habitat improved considerably over the decade since the 1995 floods (VDGIF, unpublished data); however, Brook Trout failed to naturally recolonize Garth Run or Kinsey Run. Both streams are tributaries to the Rapidan River, known as Virginia's premiere wild Brook Trout destination for anglers. Another tributary of the Rapidan River impacted by debris flows was the Staunton River. Proper conditions existed in the Staunton River for recolonization including a nearby source population (extreme headwaters unaffected by debris flows), lack of physical barriers, and suitable habitat in the area affected (Roghair and Dolloff 2005). Post-event snorkel surveys verified Brook Trout recolonization of the impacted area at a rate of several hundred meters per year (Roghair and Dolloff 2005). However, Brook Trout failed to recolonize Garth Run and Kinsey Run, apparently due to the lack of a nearby source population. VDGIF biologists made the decision in 2008 to reintroduce wild Brook Trout into Garth Run and Kinsey Run with the goal of developing self-sustaining populations where historical populations once resided.



Figure 1. Aerial view of debris flow and flood deposits from the June 27, 1995 flood at Kinsey Run, near Graves Mill in Madison County, Virginia. Photo from “The debris flows of Madison County, Virginia: 34th Annual Virginia Geological Field Conference Guidebook”.

Methods

Garth Run (elevation 345–486 m) and Kinsey Run (elevation 291–440 m) are second order tributaries to the Rapidan River in Madison County within the Rapidan Wildlife Management Area (WMA) (Figure 2).

On September 10, 2008, wild Brook Trout were collected by backpack electrofishing from the Conway River, a neighboring watershed on the Rapidan WMA, for stocking into Garth Run and Kinsey Run. The Conway River was chosen as the donor stream due to its proximity to the reintroduction streams (<2.5 km), presumed similar genetics, and ease of access. September was selected as the month of reintroduction to accommodate potential fall spawning. All trout collected for reintroduction were measured (TL) and weighed (g) prior to loading on a pickup truck fitted with an insulated stocking tank with two agitators. Garth Run received 107 trout ranging from 61 to 204

mm TL (mean = 121 mm, SD = 34), while Kinsey Run received 104 trout ranging from 52 to 210 mm TL (mean = 117 mm, SD = 38) (Figure 3). There was no significant difference in mean length of fish stocked between streams (ANOVA; $P = 0.75$). A representative size distribution was collected for transplantation from the Conway River to mimic natural population size structure found in area streams.

Multiple-pass removal techniques are generally used to estimate trout abundance (Riley and Fausch 1992; Kruse et al. 1998). However, due to time constraints, personnel scheduling, and to reduce the chance of sampling mortality; single-pass electrofishing surveys were chosen as our sampling method. Bateman et al. (2005) determined that single-pass sampling captured 74 to 78% of a trout population and was effective in determining spatial patterns and abundance. Kruse et al. (1998) also found that single-pass electrofishing accurately displayed abundance

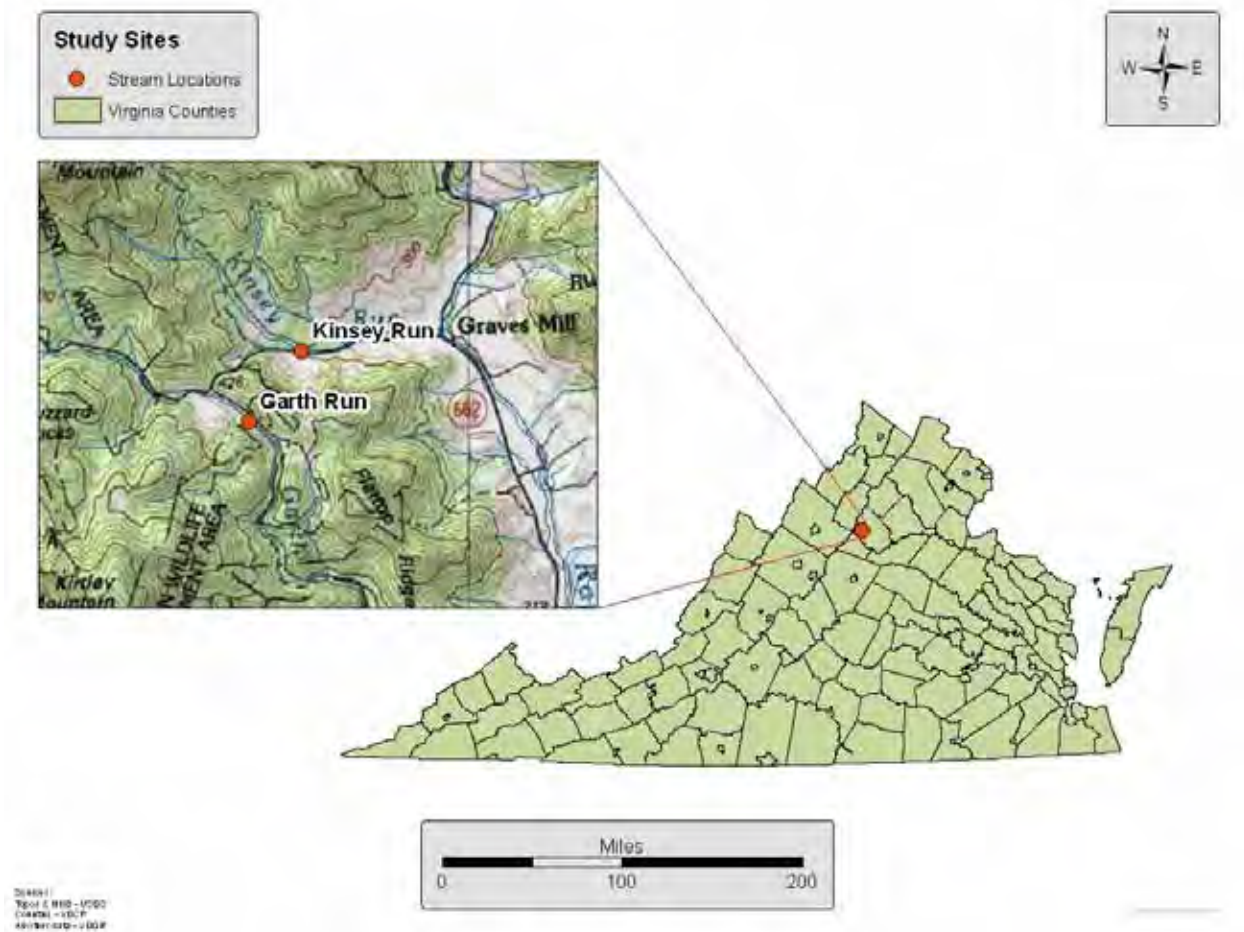


Figure 2. Study streams impacted by catastrophic flooding associated with the June 27, 1995 Madison County, Virginia floods that were selected for Brook Trout reintroductions.

of trout in small headwater streams similar to ours. Single-pass electrofishing surveys were conducted annually from June 2009 to 2016 to monitor survival and reproduction (Kinsey Run was not sampled in 2012). Brook Trout collected were measured (TL) and weighed (g) prior to release, and trout numbers were tracked as fish per 100 m. Brook Trout >100 mm TL were considered adults, and fish <100 mm were assumed to be young of the year (YOY) (Fausch and White 1981; Marschall and Crowder 1996).

Results

Brook Trout reproduction was documented in both Garth Run and Kinsey Run during the first survey in June 2009 – roughly 9 months post reintroduction (Table 1). Reproduction from 2009 to 2016 was variable (Table 1) and mimicked trends documented

in regional streams. However, recruitment variability was not significant among years, but was significant between streams (ANOVA, $P = 0.001$).

Adult Brook Trout numbers were also variable, as initial catch rates declined post introduction and then began to expand 7 years later (Table 1). Catch rates for both YOY and adult Brook Trout from Kinsey Run were significantly higher throughout the survey compared to Garth Run (Table 1). Both streams had increasing populations 8 years post introduction (Figures 4, 5), but expansion rates varied. Population indices documented a diverse size frequency; however, Garth Run consistently produced the largest trout with specimens up to 30 cm (Figure 4). Brook Trout size structure and recruitment were more stable in Kinsey Run, but adult trout never exceeded 22 cm TL (Figure 5).

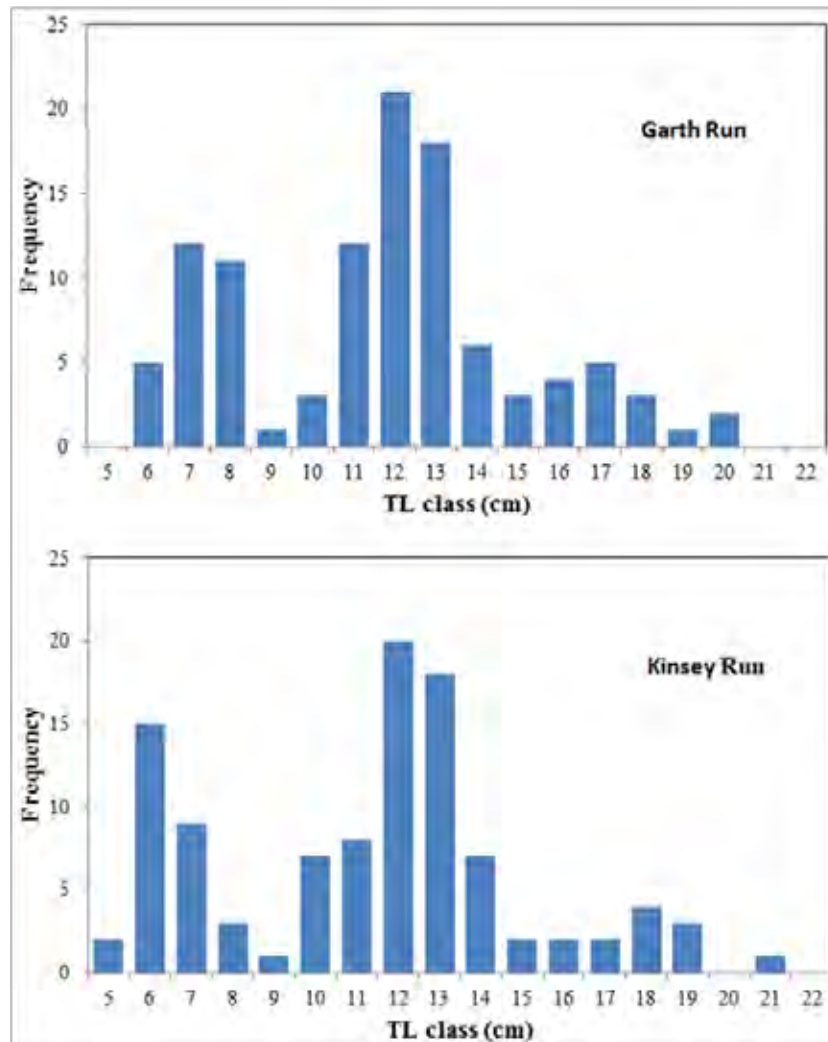


Figure 3. Brook Trout length frequency data (cm class) for fish reintroduced into Garth Run and Kinsey Run, Virginia. Brook Trout were collected on September 10, 2008 from the Conway River, Virginia; which is an adjacent watershed.

Table 1. Brook Trout catch rates per 100 meters of stream surveyed, 2009-2016. No datum is denoted as ND.

		June 2009	June 2010	June 2011	June 2012	June 2013	June 2014	June 2015	June 2016
Garth Run	Adult	0	4.3	0.7	1.0	9.1	3.0	13.7	19.0
	YOY	1.7	0	0	1.0	0	1.3	4	3
	Total	1.7	4.3	0.7	2.0	9.1	4.3	17.7	22
Kinsey Run	Adult	1.8	6	3.9	ND	34.0	30.0	45.0	66.0
	YOY	11.3	2.1	11.3	ND	32.0	16.0	20.0	12.0
	Total	13.1	8.1	15.2	ND	66.0	46.0	65.0	78.0

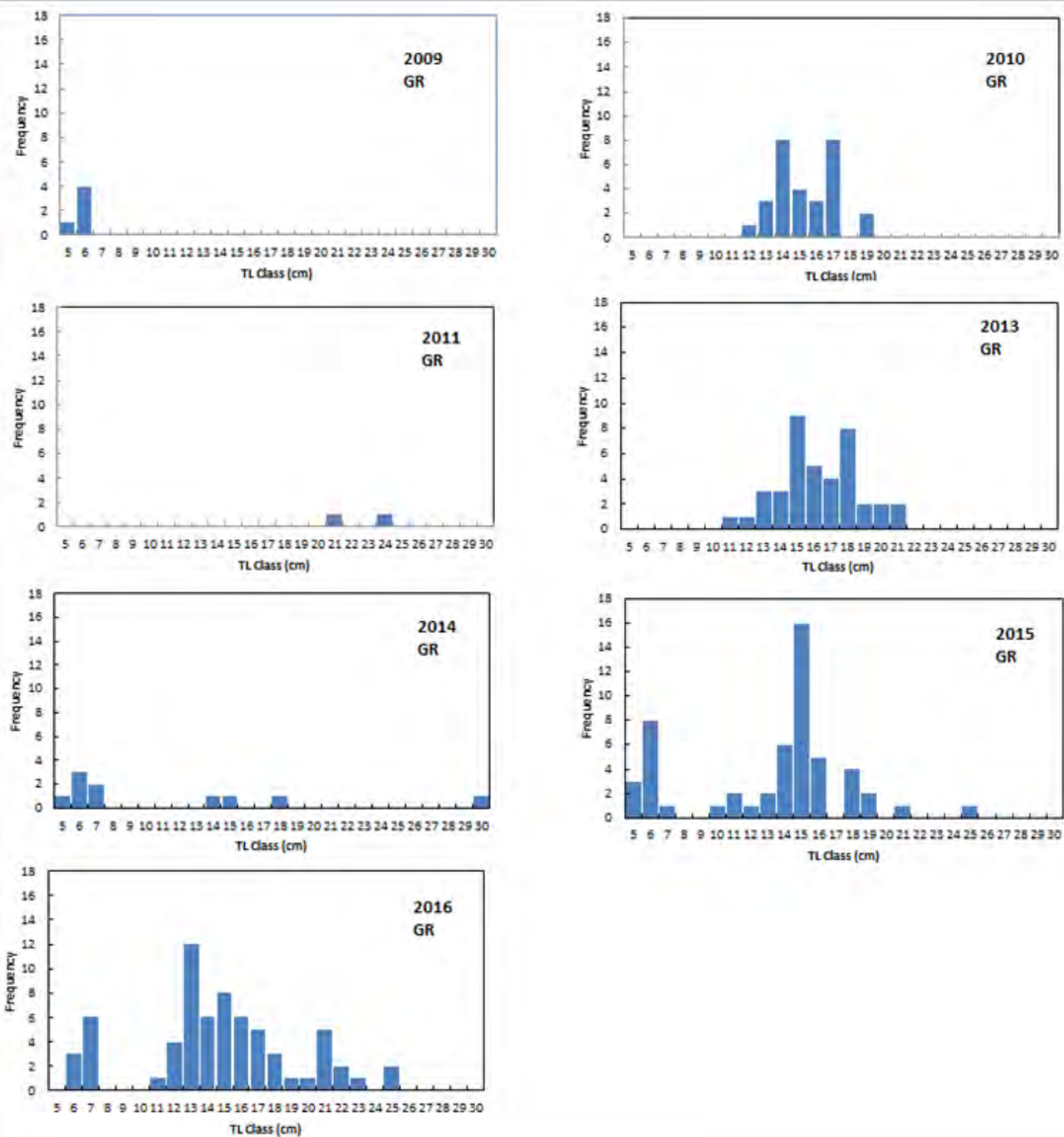


Figure 4. Brook Trout length frequency data for Garth Run (GR), Virginia, up to 8 years post reintroduction (June 2016).

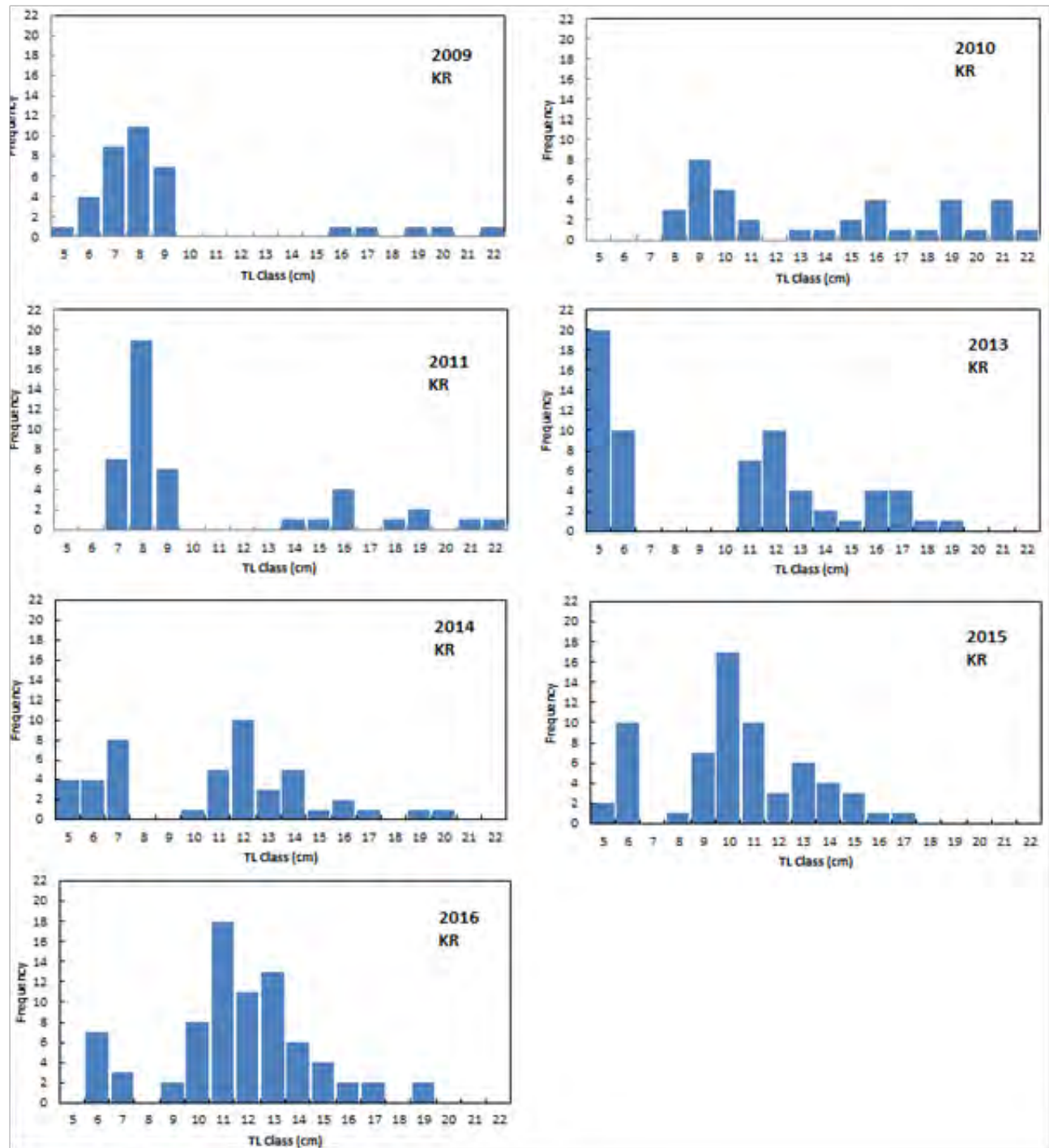


Figure 5. Brook Trout length frequency data for Kinsey Run (KR), Virginia, up to 8 years post reintroduction (June 2016).

Discussion

Our study demonstrated that successful Brook Trout reintroduction could occur with approximately 100 fish of various sizes collected from a neighboring watershed transplanted in early fall. Newly transplanted adults were able to spawn in study streams during the first few months of residency, and YOY were collected the following June in both waters. It appears that it may not be necessary to transplant large numbers of trout into small headwater streams to establish reproducing populations when acceptable water quality and spawning habitat are present. This is an important finding, as previous studies have described the failure of hatchery-reared fingerling trout stocked in the wild from developing reproducing populations (Potter and Barton 1986). Genetic concerns regarding the introduction of domestic hatchery stocks into watersheds containing wild trout (Marnell 1986) may be averted by relying on small, timed stockings of wild fish from within the watershed or from an adjacent or nearby watershed with similar genetic makeup.

Thus, we suggest using wild fish for reintroductions offers numerous advantages since several studies have shown stocking hatchery trout where wild trout exist may decrease wild trout abundance (Bachman 1982; Vincent 1987). Mechanisms that cause declines in wild trout numbers after hatchery trout are stocked are not fully understood, but a disruption of existing social behavior may be a factor (Vincent 1987; Elinum and Fleming 2001). McLaren (1979) found the activity of wild trout increased following the introduction of hatchery trout. This may alter feeding strategies, increase movement, or increase aggression which may lead to stresses that can reduce the stream's natural carrying capacity (Vincent 1987; Elinum and Fleming 2001). Problems which alter social hierarchy may be limited by translocation of wild fish from a donor stream. Impacts to donor streams can also be minimized by removing as few fish as possible for transplantation into other streams while still providing enough fish for reproductive success into newly occupied habitats. Other studies have found evidence suggesting minimal or no effects from stocking hatchery trout into systems with existing wild trout populations (Meyer et al. 2012; Schill 2014). Therefore, lacking conclusive findings our design focused on wild trout

introductions. Information is lacking as it pertains to stocking wild fish into suitable new waters or restoration waters as well as recommendations pertaining to stocking densities that may lead to successful recruitment. In our study, approximately 100 Brook Trout per stream met that threshold of fish needed for reproduction and recruitment.

Future management efforts to reintroduce trout into streams that formerly supported salmonids or into new systems that have acceptable water quality and habitat will focus on utilizing small stockings of wild trout collected from donor streams with similar habitat and water quality.

Acknowledgments

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Science to Action: Decision-support to Advance Stream Trout Management in a Changing Climate

Andrew K. Carlson^{1*}, William W. Taylor², Zeenatul Basher³,
T. Douglas Beard, Jr.⁴, Dana M. Infante⁵

¹Ph.D. Student, ²Distinguished Professor, ³Research Scientist, ⁴Chief, National Climate Change and Wildlife Science Center, US Geological Survey, ⁵Associate Professor, Center for Systems Integration and Sustainability, Department of Fisheries and Wildlife, Michigan State University

Abstract—Projected increases in coldwater stream temperatures resulting from predicted air temperature warming over the next 50 years are cause for concern among fisheries professionals that manage Brook Trout *Salvelinus fontinalis*, Brown Trout *Salmo trutta*, and Rainbow Trout *Oncorhynchus mykiss*. We collaborated with Michigan fisheries professionals to coproduce a decision-support tool to facilitate management decision-making in 52 trout streams amid climatic changes. The tool ranks streams based on manager-defined stream criteria (e.g., current and projected 2056 temperature, groundwater contribution, trout relative abundance, watershed and riparian land cover), enabling fisheries professionals to make ecologically, socioeconomically robust management decisions that promote thermally resilient streams and trout populations. Stream ranking using all criteria indicated that certain recreationally significant fisheries (e.g., Muskegon River) will experience warming that may cause them to become less important for trout management. However, lesser-known fisheries (e.g., Davenport Creek) were projected to become more thermally suitable and important for trout management. With this information available, managers can anticipate future thermal, hydrological, and biological conditions in streams and thereby make informed, resilience-based management decisions to sustain trout fisheries. Our research demonstrates the utility of synthesizing multiple information sources to facilitate efficient, effective decision-making amid complex fisheries management environments in a changing climate.

Introduction

Fisheries professionals are often faced with difficult policy and management decisions that require them to synthesize biological, social, economic, and political information to create productive, sustainable fisheries. With such interconnected factors to consider, fisheries professionals would benefit from methods that integrate different information sources to enable management decision-making that is adaptive, inclusive, and efficient (Lynch et al. 2015). Decision-support tools (DSTs) – information systems that organizations and individuals use to make science-based decisions – facilitate decision-making by systematically integrating diverse human and environmental information and allowing efficient evaluation of trade-offs among possible options (NRC 2010). A promising decision-support application is to use DSTs to inform fisheries management in a changing climate, which will exert socio-ecological consequences that impact fisheries at the individual, population, community, and ecosystem levels (Paukert

et al. 2016; Carlson et al. 2017). In particular, climate warming reduces recruitment, growth, abundance, and distribution of coldwater fish in families Salmonidae and Cottidae (Lyons et al. 2009) and impacts fisheries stakeholders (e.g., recreational fishers, fisheries professionals) through its effects on fish populations, habitats, and environmental conditions (Paukert et al. 2016).

In recent years, advancements in Internet communication and computing power have caused decision support science to expand rapidly into disciplines such as fisheries where it was traditionally absent. Although fisheries decision support is a relatively nascent field, it has numerous applications with the potential to advance fisheries management (Azadivar et al. 2009; Bitunjac et al. 2016). Decision-support tools (DSTs) facilitate information synthesis such that fisheries professionals can use them to rapidly model the wide-ranging ecological, social, economic, and policy conditions they encounter and thereby facilitate informed management decision-

making. In addition, fisheries professionals can make fisheries management more transparent, objective, and socially acceptable if they engage public stakeholders as DSTs are developed and used (NRC 2010). Moreover, the design and implementation of DSTs often requires co-production (i.e., collaboration among scientists and science users to make policy and management decisions) to develop effective instruments for climate change adaptation (Meadow et al. 2015). By promoting co-production among multiple fisheries and aquatic stakeholders (e.g., scientists, managers, biologists, policy-makers, anglers, general public), DSTs should provide a pathway for increasing the resilience of fisheries and fisheries management

systems (i.e., their ability to absorb disturbance and retain structure and function amidst climate change; Holling 1973). Such resilience-based management is imperative as climate change continues to affect fisheries throughout the world (Paukert et al. 2016; Carlson et al. 2017). The goal of this study was to design and implement a map-based DST to assist fisheries professionals in planning management programs that promote resilient coldwater streams, trout populations, and associated human systems. Specific objectives were to: (1) coproduce a DST through an academic-agency-public partnership; (2) assess the implications of DSTs for resilience-based stream fisheries management within and outside Michigan.

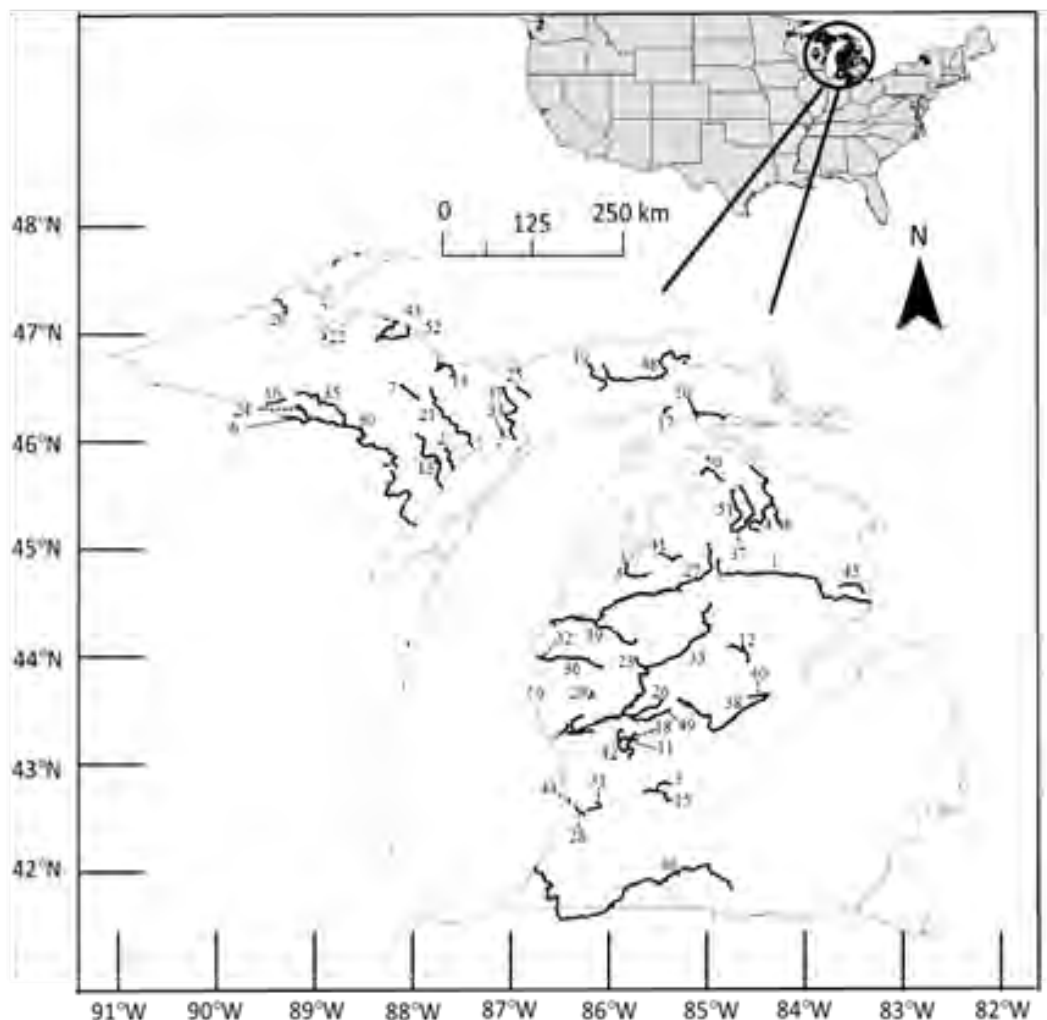


Figure 1. Map of 52 Brook Trout, Brown Trout, and Rainbow Trout streams in Michigan used for air-stream temperature modeling and decision-support tool development. Streams identification numbers and trout species information are available in the online decision-support tool (<https://goo.gl/pM1ug7>) and Table 1 of Carlson et al. (2017).

Methods

Study Area

We selected 52 Michigan streams across a latitudinal gradient (approximately 47° N to 42° N; Figure 1) to ensure variation in current and projected future air and stream temperatures. We also chose streams to span a gradient from groundwater to surface-runoff dominance to encompass the hydrological and thermal conditions that Brook Trout *Salvelinus fontinalis*, Brown Trout *Salmo trutta*, and Rainbow Trout *Oncorhynchus mykiss* experience in watersheds throughout Michigan (Carlson et al. 2017). All streams had recreational fisheries for one or more of these species; thus, they were important for public stakeholders and Michigan fisheries professionals.

Fisheries Professional Survey

We designed a 30-question survey instrument to evaluate the opinions and perspectives of Michigan fisheries professionals regarding trout management in a changing climate. The survey was approved by the Michigan State University Institutional Review Board (IRB # x16-1438e; i052807) and delivered to 40 Michigan Department of Natural Resources (MDNR) fisheries professionals (23% of fisheries staff) via SurveyMonkey®, with reminder emails sent every 3 weeks during a 2.5-month time span from November 2016 to February 2017 in which the survey was open. In total, 31 fisheries professionals participated in the survey (78% response rate). We designed survey questions in consultation with survey specialists from Michigan State University to

ensure that they were succinct, yet detailed enough to provide necessary thermal, hydrological, biological, and socioeconomic information for stream trout management and DST development. The survey encompassed a range of questions designed to assess the perspectives of fisheries professionals regarding trout thermal habitat management in Michigan and ways in which resource availability (e.g., money, time, personnel) influences their decisions regarding trout management strategies. Fisheries professionals were also asked to rank the relative importance of thermal, hydrological, biological, and socioeconomic factors, which were subsequently used to define DST criteria (Table 1) that define critical environmental conditions for sustaining thermally resilient coldwater streams and trout populations: water temperature, trout population characteristics (i.e., presence/absence, relative abundance), groundwater contribution to streamflow, watershed land cover, riparian land cover, and projected 2056 water temperature (Carlson et al. 2017).

Stream Temperature

We obtained daily water temperatures (1990–2010) for all streams from an MDNR database. Because headwaters of Michigan streams are generally coolest and most optimal for trout in summer (Drake and Taylor 1996), we used temperature gauges from each stream’s headwaters. We used the North American Anthropogenic Barrier Dataset (Ostroff et al. 2013) to locate and omit stream temperature gauges directly below dams, which have been shown to artificially increase temperatures (Lessard 2000). We used mean daily water temperatures in July to

Table 1. Six decision-support tool criteria that define critical environmental conditions for sustaining thermally resilient coldwater streams and trout populations. Criteria were used to rank streams according to their importance for trout management. Weights are manager-defined relative importance values for the six criteria according to responses from the survey of fisheries professionals.

Criterion	Source	Weight
Water temperature (°C)	MDNR	0.23
Trout presence/absence, CPUE	MDNR (Wills et al. 2015)	0.20
Groundwater contribution (baseflow index)	USGS (Neff et al. 2005)	0.17
Watershed land cover (%)	NLCD 2011 (Homer et al. 2015)	0.14
Riparian land cover (%)	NLCD 2011 (Homer et al. 2015)	0.14
Projected 2056 water temperature (°C)	Carlson et al. 2016, 2017	0.11

define the water temperature criterion for the DST (Table 1), because this is the time period that generally has the warmest Michigan stream temperatures and thus is the least optimal for trout growth and survival (Carlson et al. 2017). We used temperatures measured in July 2006 for the DST water temperature criterion as MDNR records for this year were the most spatially extensive and complete.

We also used predicted changes in stream-specific water temperatures as a DST criterion. We projected stream temperatures in the year 2056 by inputting future air temperature predictions into air-water temperature regression models for each stream (see Carlson et al. 2017). For the 28 streams for which historical stream temperatures were available, we used stream-specific regression models to predict water temperatures in 2056. If streams did not have historical water temperatures, we used generalized (i.e., region-specific) air-water temperature models developed for surface runoff-dominated (Stefan and Preud'homme 1993) and groundwater-dominated (Krider et al. 2013) streams.

Trout Population Characteristics

We obtained data on the presence/absence and relative abundance (i.e., catch per unit effort; number of fish/mile) of Brook Trout, Brown Trout, and Rainbow Trout in each of the 52 streams from MDNR standardized electrofishing surveys (Wills et al. 2015). We converted relative abundance to a categorical measurement as defined by the MDNR: high abundance (>75th percentile), medium abundance (25th-75th percentile), low abundance (< 25th percentile; Wills et al. 2015).

Groundwater Contribution

We used the contribution of groundwater to streamflow as a DST criterion, because groundwater provides temperature buffering and flow stability that typically make streams more thermally suitable for trout than surface-runoff dominated systems. We represented stream-specific groundwater inputs using baseflow index (BFI) values, which express mean baseflow (mm·year⁻¹) divided by total streamflow (mm·year⁻¹; Neff et al. 2005). We defined groundwater-dominated streams as those with BFI > 0.60, whereas surface runoff-dominated systems were those with BFI ≤ 0.60 (Dukić and Mihailović 2012).

Watershed and Riparian Land Cover

We also used watershed and riparian land cover as input criteria for the DST due to their importance in determining physical and thermal habitat quality and quantity for stream trout (Siitari et al. 2011). We evaluated watershed and riparian land cover using the 2011 National Land Cover Database (Homer et al. 2015). We used four land cover types that promote high-quality thermal habitats for trout (i.e., stream shading, high groundwater recharge) to define the watershed and riparian land cover criteria of the DST: deciduous forest, evergreen forest, mixed forest, and grassland (Siitari et al. 2011). We defined riparian zone width as 100 m to be consistent with previous trout stream research (Carlson et al. 2016).

Decision-Support Tool Design and Delivery

We used results from the fisheries professional survey to define weights (i.e., relative importance values) for each DST criterion (Table 1). Higher weights corresponded to greater importance, with the sum of all criteria weights equaling one. We ranked each stream's thermal, hydrological, and biological characteristics – as measured by the six DST criteria – relative to those of other streams by calculating stream-specific output scores for all criteria. We calculated output scores such that higher scores corresponded with higher-quality thermal habitat conditions or, for the trout population characteristics criterion, higher-quality trout populations (i.e., more species present and/or greater relative abundance; Table 1). Although BFI values were proportional to trout thermal habitat quality (i.e., higher BFI values corresponded with higher-quality habitat conditions), water temperatures in 2006 and 2056 had to be adjusted so that lower temperatures produced higher scores, and vice versa. To perform this adjustment, we subtracted each stream temperature from 25 °C – a value above the water temperatures measured and projected herein – so that cooler streams temperatures received higher scores and warmer streams received lower scores. We assigned scores for the trout population characteristics criterion in binary format for the presence (score = 1) and absence (score = 0) of particular species (i.e., Brook Trout, Brown Trout, Rainbow Trout) and increasing numerical format for low relative abundance (score = 1), medium relative

abundance (score = 2), and high relative abundance (score = 3) as defined by Wills et al. (2015). We added scores across species such that streams with Brook Trout, Brown Trout, and Rainbow Trout in high abundance received higher scores than streams with low abundance or absent trout species. The cumulative percentage of each stream's watershed and

riparian zone composed of high-quality cover types (i.e., deciduous forest, evergreen forest, mixed forest, grassland) represented scores for the watershed and riparian land cover criteria, respectively.

For each stream, we normalized output scores for each input criterion to a consistent scale of 0 to 100 using the following formula:

$$\frac{[\text{individual stream score}] - [\text{minimum score (all streams)}]}{[\text{maximum score (all streams)}] - [\text{minimum score (all streams)}]} \times 100$$

We performed this calculation individually for each input criterion, and we multiplied normalized scores by the reported manager-defined weights (Table 1) to produce weighted scores (Rohweder et al. 2015). We then added six weighted scores for each stream to produce a final stream score that reflected all DST criteria. We ranked streams by final stream scores, which represented each stream's overall importance for trout management relative to other streams. We designed and delivered the DST to Michigan fisheries professionals using Data Basin®, an open-access, user-friendly mapping and analysis platform. We assembled spatial data representing each of the six input criteria in ArcMap 10.4 (Environmental Systems Research Institute, Redlands, California) and exported them to Data Basin, which we configured to display six map layers, one for each criterion. We displayed final stream scores reflecting all six criteria in an additional map layer that depicted each stream's overall importance for trout management. The DST is available open-access at <https://goo.gl/pM1ug7>.

Results

Stream Temperature

Davenport Creek (eastern Upper Peninsula; Figure 1) had the coolest July water temperature (11.5°C) and was thus the most thermally optimal trout stream studied herein. Other highly-ranked streams in terms of water temperature were the West Branch Sturgeon River (13.3°C, northern Lower Peninsula) and the Manistee River (13.6°C, northern Lower Peninsula; Figure 1). Davenport Creek and the West Branch Sturgeon and Manistee rivers were the highest-ranking streams for the projected future water temperature criterion, with projected mean July water temperatures in the year 2056 that were ≤ 14.2°C.

Trout Population Characteristics

The Au Sable, Manistee, and West Branch Sturgeon rivers in Michigan's northern Lower Peninsula (Figure 1) received the highest rankings for the trout population characteristics criterion. These are three of Michigan's premier trout streams (MDNR 2017) as they all have Brook Trout, Brown Trout, and Rainbow Trout in relatively high abundance and size.

Groundwater Contribution

The Ogontz and Sturgeon rivers (central Upper Peninsula) were the highest-ranking streams with respect to groundwater contribution. Five other trout streams (i.e., East Branch Fox and Little Indian rivers, central Upper Peninsula; Au Sable and South Branch Pine rivers, northern Lower Peninsula; Carlton Creek; Figure 1) had BFI values > 0.70, meaning they are thermally buffered and are likely to maintain optimal or suitable thermal habitat conditions in a changing climate.

Watershed and Riparian Land Cover

Davenport Creek and the Salmon Trout River (western Upper Peninsula, Figure 1) were ranked highest for watershed land cover. Other high-ranking streams for the watershed land cover criterion included Mann Creek (southern Lower Peninsula), the Falls River (western Upper Peninsula, Figure 1), and the East Branch Fox River, each with 82.6–87.2% of their riparian zones composed of deciduous forest, evergreen forest, mixed forest, or grassland. The East Branch Fox River was the highest-ranking stream for the riparian land cover, followed by the Elm River (western Upper Peninsula), Bryan Creek (central Upper Peninsula, Figure 1), and Mann Creek.

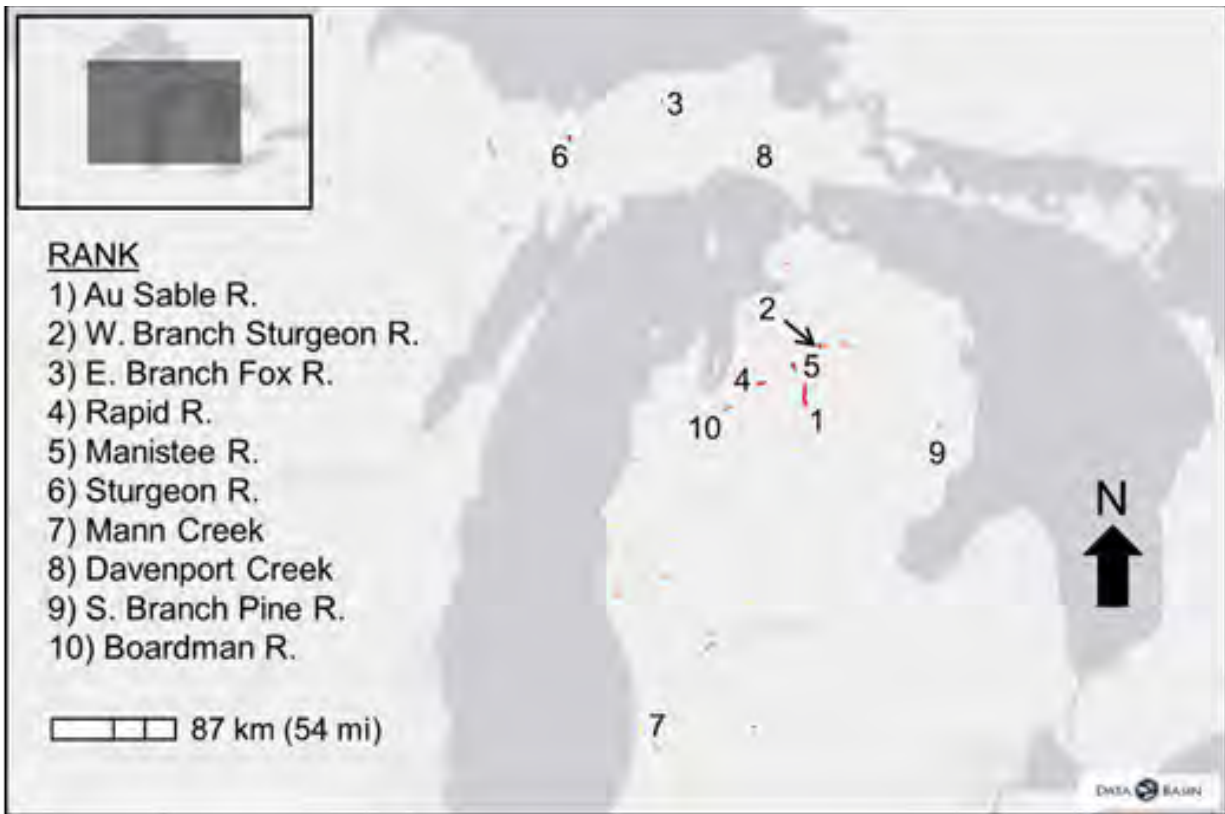


Figure 2. Michigan streams ranked by importance for trout management based on six decision-support tool criteria (Table 1). Rankings were assigned by adding weighted, criterion-specific scores for each stream and arranging final scores from largest to smallest.

Considering all six criteria simultaneously, the highest-ranking streams (i.e., greatest importance for trout management) were the Au Sable, West Branch Sturgeon, East Branch Fox, Rapid, and Manistee rivers (Figure 2).

Discussion

Using a case study of Michigan trout streams, this study demonstrates the utility of DSTs for facilitating fisheries decision-making amid complex biological, social, economic, and political conditions, adding to a relatively sparse literature on value of fisheries decision-support for resilience-based management (Azadivar et al. 2009; Bitunjac et al. 2016). Whereas previous studies generally used DSTs in the context of commercial fisheries management in marine systems, we demonstrated the utility of DSTs for informing freshwater recreational fisheries management in a changing climate. The present study illustrates how co-production between fisheries professionals and

stakeholders (e.g., anglers, policy-makers, general public) is useful for developing DSTs that promote informed, anticipatory fisheries decision-making.

Some of the findings generated by the stream trout DST were expected, whereas others were unforeseen. For instance, the Au Sable River supports a renowned trout fishery and has been named a National Wild and Scenic River and a Michigan Blue Ribbon Trout Stream (Canale and Chapra 2016), so it is not surprising that it was the highest-ranked stream. It was also foreseeable that a majority of the ten highest-ranking trout streams in this study were located in the Northern Lower Peninsula (NLP) of Michigan, where geological, climatic, and hydrological conditions are currently favorable for productive stream trout fisheries (T. Zorn, MDNR, personal communication). In contrast, certain socioeconomically important fisheries (e.g., Muskegon River, Pere Marquette River) were predicted to experience thermal habitat degradation that may cause them to become less

important for trout management than at present, whereas lesser-known fisheries (e.g., Rapid River, Davenport Creek) were projected to become more important for trout management. These findings demonstrate how DSTs can reveal surprising, management-relevant information and represent flexible instruments for fisheries professionals as they make decisions on how to best sustain stream trout fisheries amid climatic and environmental changes. Rather than prescribe trout management actions in specific streams, a focus of this study was to provide fisheries professionals with information for their own decision-making. In accordance with the eight steps of the fisheries management process (Taylor et al. 1995), stream-specific conditions (e.g., physical, chemical, biological, social) will determine how fisheries professionals use the information DSTs supply. For example, fisheries professionals and stakeholders may consider high-importance streams *high-priority* systems for habitat protection or rehabilitation under certain circumstances (e.g., trophy fisheries management, threatened/endangered species conservation) but *low-priority* systems for these activities in other situations (e.g., lower-ranking streams that are more vulnerable to climate change effects are deemed worthy of protection/rehabilitation). The DST is a flexible instrument for stream trout management because it informs, rather than makes, management decisions.

The stream trout DST is a mechanism for resilience-based fisheries management because it allows fisheries professionals to predict future thermal habitat conditions and use this information to proactively manage trout populations and their habitats for resilience in a changing climate. Along with climate-induced changes to fisheries ecosystems, the human components of fisheries systems (e.g., fisheries professionals, commercial and recreational fishers) are directly affected by alterations in fish abundance and distribution resulting from climate change (Paukert et al. 2016). Thus, because both the natural and human components of fisheries systems are changing as the climate warms, it is important to manage fish, their habitats, and allied human resources for resilience (Paukert et al. 2016). Resilience-based management involves collaboration among professional and non-professional fisheries stakeholders and consideration of ecological, socioeconomic, and political information to facilitate robust management strategies

(Carlson et al. 2017). As demonstrated herein, DSTs can be valuable instruments for resilience-based management because they arise from co-production and synthesize information about ecosystems and human systems, enabling fisheries professionals to design ecologically and socioeconomically informed management strategies that sustain coldwater streams and trout populations.

In summary, we found that DSTs have the potential to be useful for predicting the impacts of climate change in Michigan trout streams (e.g., magnitude and spatiotemporal distribution of warming), evaluating the need for and appropriateness of fisheries management actions (e.g., thermal habitat management, fish stocking), and prioritizing future habitat management and rehabilitation efforts in multiple watersheds. By integrating diverse environmental, social, economic, and political information to facilitate decision-making in ways that traditional fisheries management strategies generally do not, DSTs allow fisheries professionals and stakeholders to allocate resources toward specific fisheries and management actions that most effectively promote their objectives for these systems. Fisheries stakeholders can also benefit from DSTs by acquiring fisheries knowledge, engaging with fisheries professionals in DST development and implementation, and forming partnerships with fisheries agencies to use DST results to sustain coldwater streams and trout populations. Overall, this case study demonstrates important fisheries applications of DSTs as climatic changes increase the complexity of fisheries decision-making and magnify the significance of resilience-based fisheries management.

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Fracked Fish? Effects of Unconventional Natural Gas Extraction on Wild Brook Trout

C.J. Grant¹, E.S. Hall¹, and B.E. Martin²

*Corresponding author, Grant@juniata.edu, (814) 641-6686

¹Biology Department, Juniata College, 1700 Moore St. Huntingdon, PA 16652

²Environmental Science Department, Juniata College, 1700 Moore St. Huntingdon, PA 16652

Abstract—Unconventional natural gas development and hydraulic fracturing are increasing across the United States due to global energy demands. Research has only recently begun to assess hydraulic fracturing impacts to streams, and very little study has aimed at determining effects on wild trout populations. During June and July 2012-2016, we sampled 32 remotely-located streams in Pennsylvania's Marcellus Shale basin containing Brook Trout *Salvelinus fontinalis*. At each stream, pH, fish assemblages, and Brook Trout abundance were assessed. All streams either had experienced hydraulic fracturing (F+, n=19) or not yet experienced hydraulic fracturing (F-, n=13) within their watersheds at the date of sampling. Stream pH was significantly lower at F+ sites than F- sites ($p=0.009$), and was negatively correlated with well pad density ($r=-0.495$, $p<0.001$) and well density ($r=-0.387$, $p=0.003$). Brook Trout abundance ($r=0.266$, $p=0.005$) and fish diversity ($r=0.5715$, $p<0.001$) were positively correlated with pH, while fish diversity was negatively correlated with well pad density ($r=-0.704$, $p<0.001$) and well density ($r=-0.462$, $p<0.001$). Our results suggest that hydraulic fracturing has the potential to affect stream pH, fish assemblages, and wild Brook Trout in their native habitat. Better management strategies (with respect to hydraulic fracturing) are needed to ensure the conservation of wild trout.

Introduction

Advancement of unconventional natural gas development technologies, like hydraulic fracturing, are facilitating the rapid exploitation of natural gas reserves in shale beds across the United States (Entrekin et al. 2011). The most prominent formation, the Marcellus, underlays approximately 70% of Pennsylvania and much of West Virginia, as well as parts of Maryland, and New York, overlapping with much of the native range of the wild Brook Trout *Salvelinus fontinalis*. Current estimates as to the number of wells in Pennsylvania are over 6,000 active wells with over 10,000 permitted (PADEP 2016; MCOR 2017). However, this number is expected to increase, up to a total of 60,000 wells by 2030 (Johnson 2010). Researchers have suggested that the recent rapid expansion of Marcellus shale development poses a significant threat to aquatic ecosystems in Pennsylvania (Entrekin et al. 2011; Olmstead et al. 2013).

Hydraulic fracturing and flow-back fluids contain toxic organic and inorganic chemicals such as hydrochloric acid, phenols, polycyclic aromatic hydrocarbons, and BTEX (Boswell 2011; Kharaka et al. 2013). In Pennsylvania alone, it has been estimated that over six billion liters of wastewater had

been generated by 2013 (Rahm et al. 2013), and the mismanagement of produced flow-back waters is a significant threat to our water resources (Vengosh et al. 2013). Flow-back fluids have the potential to reach streams through leaking equipment (wastewater hoses) and impoundments (evaporation pits and holding facilities), lateral blowouts and seepage, as well as backflow into the wellhead (Kargbo et al. 2010; Warner et al. 2013). Any flow-back water reaching the stream would not only directly impact water quality but may also change aquatic biodiversity.

Any perturbation from hydraulic fracturing to aquatic ecosystems has the ability to affect stream water quality, fish biodiversity, and wild trout populations. Current research efforts have shown groundwater aquifer contamination (Llewellyn et al. 2015), and surface waters have been impacted by hydraulic fracturing with changes in physiochemical properties (Entrekin et al. 2011), microbial communities (Trexler et al. 2014), macroinvertebrate biodiversity (Lutz et al. 2015), and even mercury concentrations in aquatic ecosystems (Grant et al. 2015).

Herein, we examine whether Marcellus shale natural gas exploration in northwestern Pennsylvania is affecting stream pH, fish diversity, and Brook

Trout abundance over a 5-year study period. We hypothesized that streams where hydraulic fracturing had occurred within their watershed would show decreased stream pH, fish diversity, and Brook Trout abundance. Further, we anticipated that the degree of hydraulic fracturing activity (well and well pad density) within a watershed would be negatively correlated to stream pH, fish diversity, and Brook Trout abundance.

Methods

Study Sites and Field Collection

During the period from 2012 to 2016, we sampled 32 remotely located forested watersheds containing naturally reproducing Brook Trout populations within the Marcellus shale basin in Pennsylvania (Figure 1). Data regarding drilling and hydraulic fracturing dates were compiled from the Pennsylvania Department of Environmental Protection (PADEP) well production report (2016), and from PADEP wastewater generation reports (2016) for each unique well permit within the watershed of all targeted study sites. Streams were categorized according to the presence (F+, $n=19$) or absence (F-, $n=13$) of hydraulic fracturing processes within their watersheds prior to our sampling efforts. Well and well pad densities were calculated by dividing the total number of wells by watershed surface area. Several streams changed status during the study period including Dead Man's Lick, Deer Creek, Long Run, South Branch North Fork Redbank Creek (SBNFRC), South Fork West Branch Potato Creek (SFWBPC), unnamed tributary (UNT) to East Elk Fork, UNT to Naval Hollow, UNT to West Elk Fork ($n=8$). Specific 100-m study sites were established at each stream downstream and as close as possible to the permitted well pad site(s). All watersheds were otherwise minimally disturbed with no evidence of mining legacy and few conventional oil wells and dirt roads. All sites were sampled under base flow conditions in June and July. More detailed methodologies on site selection can be found at Grant et al. (2015).

Stream pH was recorded at the centroid of flow using a Eutech PCSTestr 35 Multi-parameter test probe that was calibrated weekly. Fish assemblages were determined at each 100-m interval using a two-pass wadeable electrofishing protocol with a Smith and Root LR 24 backpack electrofisher employing a pulsed

direct current ranging from 450 to 700 V (Barbour *et al.* 1999). All fish were held until completion of second pass when all species were identified and abundances recorded. Brook Trout abundance was recorded as the total number caught from a two-pass effort.

Quantitative Analysis

Watersheds were delineated using 3.2-ft Digital Elevation Models (DEM) in ArcGIS 10.5 (PAMAP Program 2006). For each watershed, number of well pads were counted using PADEP Oil and Gas Locations layer, and was secondarily verified using waste generation reports for each pad location (PADEP 2016). In addition, total area was quantified for each watershed in ArcGIS.

Spearman's rank correlations were used to test association between well pad density, well density, stream pH, Brook Trout abundance, and fish diversity. Well and well pad density data were normalized using log transformation. Correlations were run using all data from 2012-2016. We used T-Tests to compare stream values and hydraulic fracturing status for each year and then p-values were cumulated across years using Fisher's combined probability test. This approach was used to allow for inclusion of all data across years, as status of eight streams changed (from F- to F+) during the 5-year sampling period year. All statistics were calculated in R-Studio (RStudio Team 2015) and considered significant at $\alpha=0.05$. The R package *vegan* was used to calculate fish diversities utilizing Simpson's diversity index (Oksanen et al. 2017).

Results

From 2012 to 2016 stream pH was significantly lower at F+ streams than F- streams ($p=0.009$). While Fisher's combined probability test showed significant differences in pH between F+ and F- streams across the 5-year sampling period, significant differences in pH were not observed in 2014 and 2015 when analyzed by year ($p>0.10$). Stream pH was also correlated with intensity of hydraulic fracturing activity. A significant negative correlation between well pad density and stream pH ($p<0.001$, $r=-0.495$) and well density and stream pH ($p=0.003$, $r=-0.387$) was observed for the 5-year study period (Figure 2). Relationships between stream pH and fish diversity, and stream pH and Brook Trout abundance were also observed. Across years, there was a significant positive trend between stream pH and fish diversity

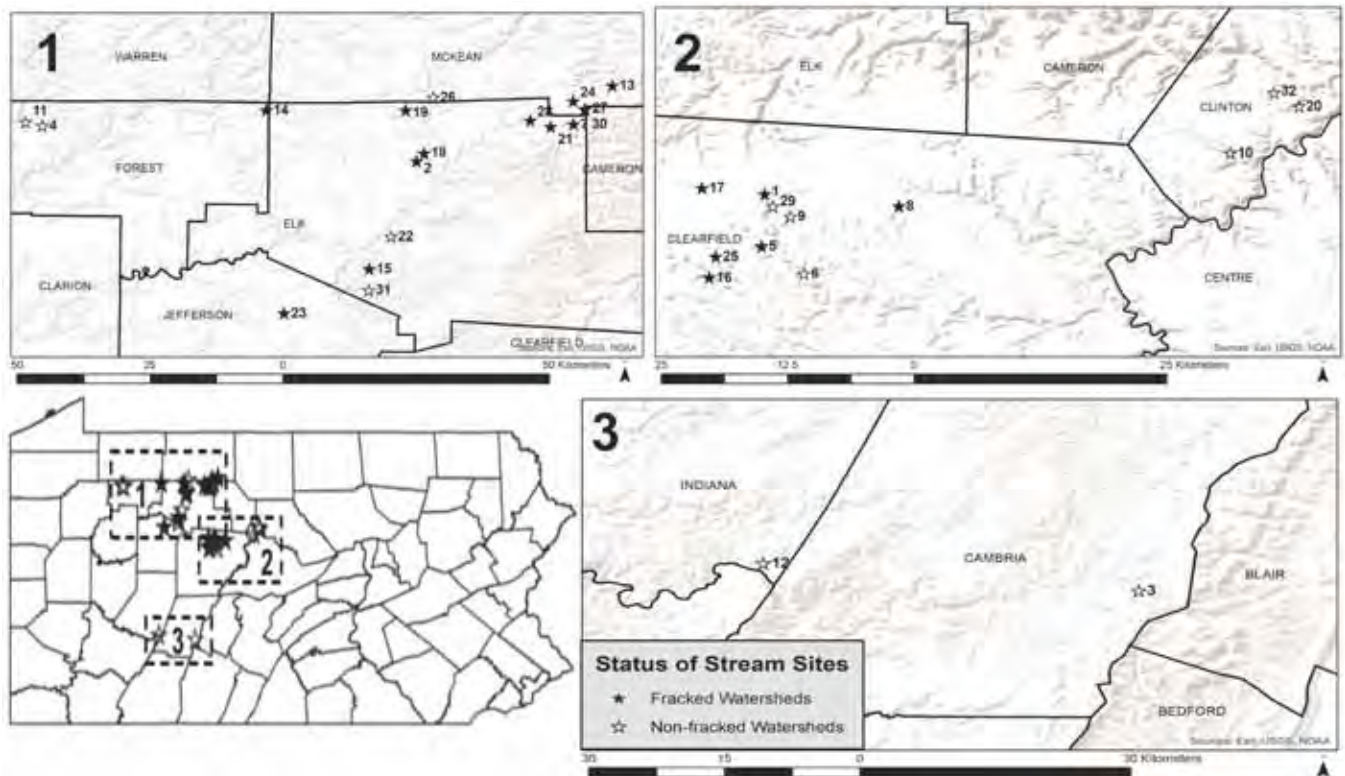


Figure 1. Map of 32 streams in Pennsylvania sampled from 2012-2016. Sites numbers correspond as follows 1-Alex Branch, 2-Bear Creek, 3-Bens Creek, 4-Camp Run, 5-Coldstream, 6-Crooked Run, 7-Dead Man's Lick, 8-Deer Creek, 9-Dixon Run, 10-Dutch Hollow, 11-East Beaver Run, 12-Findley Run, 13-Indian Run, 14-Iron Run, 15-Laurel Run, 16-Lick Run, 17-Little Laurel Run, 18-Little Wolf Run, 19-Long Run, 20-Moccasin Run, 21-Naval Hollow, 22-N. Branch Island Run, 23- SBNFRC, 24-SFWBPC, 25-Stone Run, 26-UNT Clarion River, 27-UNT to East Elk Fork, 28-UNT Naval Hollow, 29-UNT Trout Run, 30-UNT West Elk Fork, 31-Vineyard Run, 32-Wistar Run.

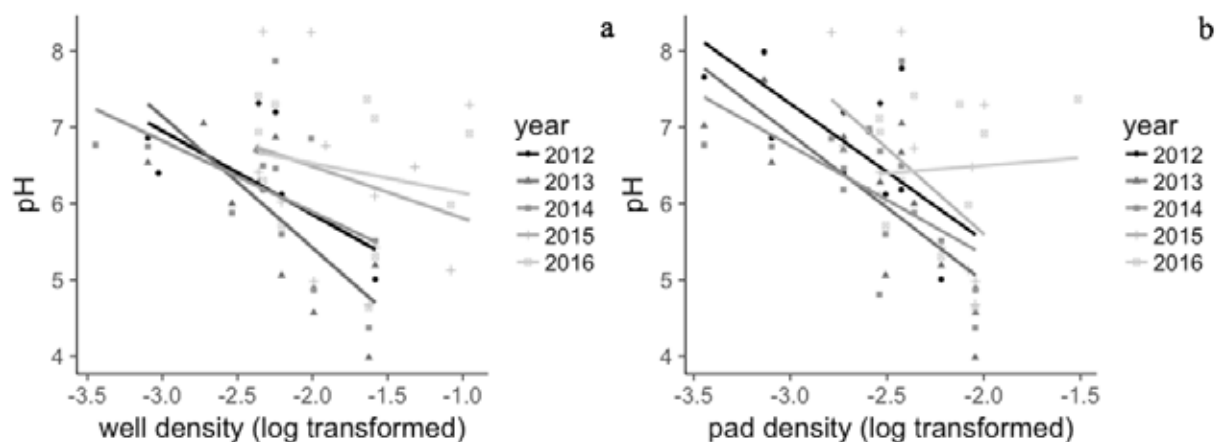


Figure 2. Stream pH levels versus within-watershed well pad density (a) and well density (b) at 32 stream sites in Northwestern Pennsylvania across a 5-year sampling effort (2012-2016). Across all years, a significant negative correlation between well pad density and pH (a) was observed ($p < 0.001$, $r = -0.4954$), as well as between well density and pH (a; $p = 0.0033$, $r = -0.3865$).

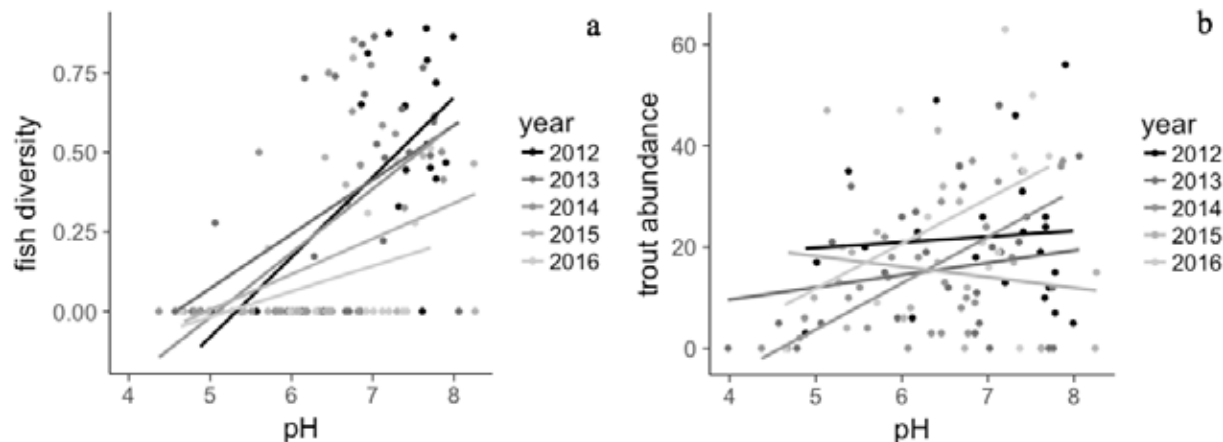


Figure 3. Stream pH levels versus fish diversity (a) and Brook Trout abundance (b) at 32 stream sites in Northwestern Pennsylvania across a 5-year sampling effort (2012-2016). Across all years, a significant positive correlation was observed for fish diversity (a; $p < 0.001$, $r = 0.5715$) and Brook Trout abundance ($p = 0.0047$, $r = 0.2663$). Stream pH ranged from 4.37 to 8.26.

(Figure 3a, $p < 0.001$, $r = 0.572$) and stream pH and Brook Trout abundance (Figure 3b, $p = 0.005$, $r = 0.266$). While fish diversity was also negatively correlated with well ($p < 0.001$, $r = -0.4618$) and well pad density ($p < 0.001$, $r = -0.7049$) at F+ sites, Brook Trout abundance was not correlated with intensity of hydraulic fracturing activity ($p > 0.10$).

Discussion

Hydraulic Fracturing and Stream pH

Decreased stream pH values at F+ streams relative to F- streams are believed to be the result of hydraulic fracturing activities. While watershed characteristics such as wetlands and conifer stands have the ability to significantly decrease stream pH (St. Louis et al. 1994), previous research on these same streams indicated no differences between such characteristics of F+ and F- streams used in this study (Grant et al. 2015; Lutz et al. 2015; Grant et al. 2016). Others have shown that hydraulic fracturing uses a number of acidic agents and solutions during the extraction of natural gasses (Kharka et al. 2013), and that these agents can reach nearby streams through leaking equipment and impoundments, lateral blowouts and seepage, as well as backflow into the wellhead (Kargbo et al. 2010; Warner et al. 2013). Well pad construction, drilling, and fracturing processes may also unearth and cross a number of

geologic formations, potentially exposing pyritic sandstone. Other research has shown that exposing pyritic sandstone can significantly affect stream and groundwater pH (Hammarstrom et al. 2013). If acidic agents are being introduced to the watershed via hydraulic fracturing activities, these agents are likely being flushed to nearby streams as a result of increased overland flow and transport.

While significant across-year differences in stream pH were observed, no significant differences were observed for the individual years of 2014 or 2015 between F+ and F- sites. We believe this may be due to an increased number of F- stream sites for those years that were not yet fracked, but had significant infrastructure developed in their watersheds including pipelines, well pads, roads, and wastewater impoundments. Further, our more continuous analysis, found both well density and well pad density in the watershed negatively correlated with stream pH across years, but well pad density was observed to have a stronger correlation than well density. This is further supported by previous findings showing that stream pH was negatively correlated with the number of well pads within a watershed (Grant et al. 2015).

While potential contamination from drilling operations and flowback fluid poses a significant threat to streams, infrastructure associated with hydraulic fracturing may be just as threatening to aquatic ecosystems. Large well pads are needed to

execute horizontal drilling and horizontal fracturing. The average Marcellus well pad in Pennsylvania is from 1.2 to 2.0 ha, but overall forest disturbance is much larger due to associated infrastructure including wastewater impoundments and spills (Johnson et al. 2010). If the collective impact of roads and pipelines are included, this area increases to about 3.5 ha of area cleared per well pad. The number of wells per pad range from one to around twelve or more, with an average of 2.15 wells per pad (Ladlee and Jacquet 2011), potentially making number of well pads a better indicator of hydraulic fracturing impact than number of wells. If these disturbances occur in predominately small forested watersheds, impacts resulting from well pad construction and supporting infrastructure (i.e. roads, pipelines, wastewater pits) have the potential to increase transport of materials from the terrestrial environments to streams.

Stream pH Effects on Fish Biodiversity and Brook Trout

Fish biodiversity appears to be largely driven by stream pH. Across all years, a significant positive correlation was observed between stream pH and fish biodiversity. Further, fish diversity was strongly negatively correlated with the density of well pads and wells within a watershed. While Brook Trout were found at most sites, they have a higher tolerance to acidic waters than other fish species (Pedder and Maly 1986; Simonin et al. 1993; Jardine et al. 2013). Eastern Blacknose Dace *Rhinichthys atratulus* and Longnosed Dace *Rhinichthys cataractae* were found in abundance at streams with zero to two well pads within their watersheds, but were not found at any stream sites with more than three well pads in the watershed where pH levels were often below six. Others have suggested that dace, unlike Brook Trout, are not tolerant of acidic waters probably because of their inability to regulate ion concentrations at low pH (Peterson et al. 1998). Overall, we observed decreased fish biodiversity and decreased numbers of pH-sensitive fish at more acidic streams that were experiencing increased drilling activities.

Results for Brook Trout abundance and stream pH did not show a strong relationship across years. While a significant positive correlation existed, a low r value was reported and variation in strength and direction was observed among years (negative correlation in 2015). The majority of streams were above the

lower pH threshold for Brook Trout (Jardine et al. 2013), and thus other unmeasured variables may be more influential on Brook Trout abundance. Also, no significant correlations were observed between Brook Trout abundance and within-watershed well and well pad density. However, low stream pH and hydraulic fracturing development may have an impact on Brook Trout food source over time, as research has shown acidic stream water and increased sedimentation can negatively impact macroinvertebrate communities (Jonsson et al. 2017). It was observed that streams experiencing pH below the lower Brook Trout threshold or streams with highest within-watershed well pad densities, only 2 or less individuals were captured in a blocked 2-pass 100-m electrofishing effort.

Impacts of Spills on Fish Diversity and Brook Trout Abundance

Streams sites with the highest intensity of drilling and documented wastewater spills had the largest decreases in stream pH, fish diversity, and Brook Trout abundance. No fish were found at Little Laurel Run which has been historically classified as a high quality coldwater fishery sustained by naturally reproducing Brook Trout populations. The PA Department of Environmental Protection documented a well blowout with hydraulic fracturing fluid reaching the Little Laurel Run in 2010 that resulted in decreased water quality, fish kills (Levis 2011), and no subsequent repopulation of fishes by the completion of our sampling in 2016. Further, ongoing contamination of Alex Branch between 2009 and 2010 resulted in surfactants, flowback fluid, and well-wash fluid negatively impacting the stream ecosystem (Levis 2011). These spills into Alex Branch impacted fish communities, resulting in low Brook Trout numbers (<5 individuals captured/year) and no occurrence of Eastern Blacknose Dace and Longnose Dace during our sampling. Alex Branch and Little Laurel were also the two most acidic streams in this study, and previous research showed that macroinvertebrate scrapers were absent from Little Laurel Run (Lutz et al. 2015). It is commonly believed that if aquatic communities become impaired as a result of low pH, more specialized macroinvertebrate feeders (e.g. scrapers) would be the first to disappear (Barbour et al. 1998). Further, lowered stream pH increases solubility of other contaminants, potentially leading to increased

bioaccumulation of toxins in higher trophic order fish (Jardine et al. 2013; Grant et al. 2016).

From this study, drilling activities appear to have an impact on stream pH, fish biodiversity, and Brook Trout abundance. To further elucidate findings and clarify pathways of contamination, future work should focus on sampling the same streams before and after hydraulic fracturing, as well as measuring streams at various stages of development (i.e. road construction, well pad development, pipeline connection). The level of detail gained by this future work, when coupled with current findings, should help implement better management practices for hydraulic fracturing activities in close proximity to wild Brook Trout streams.

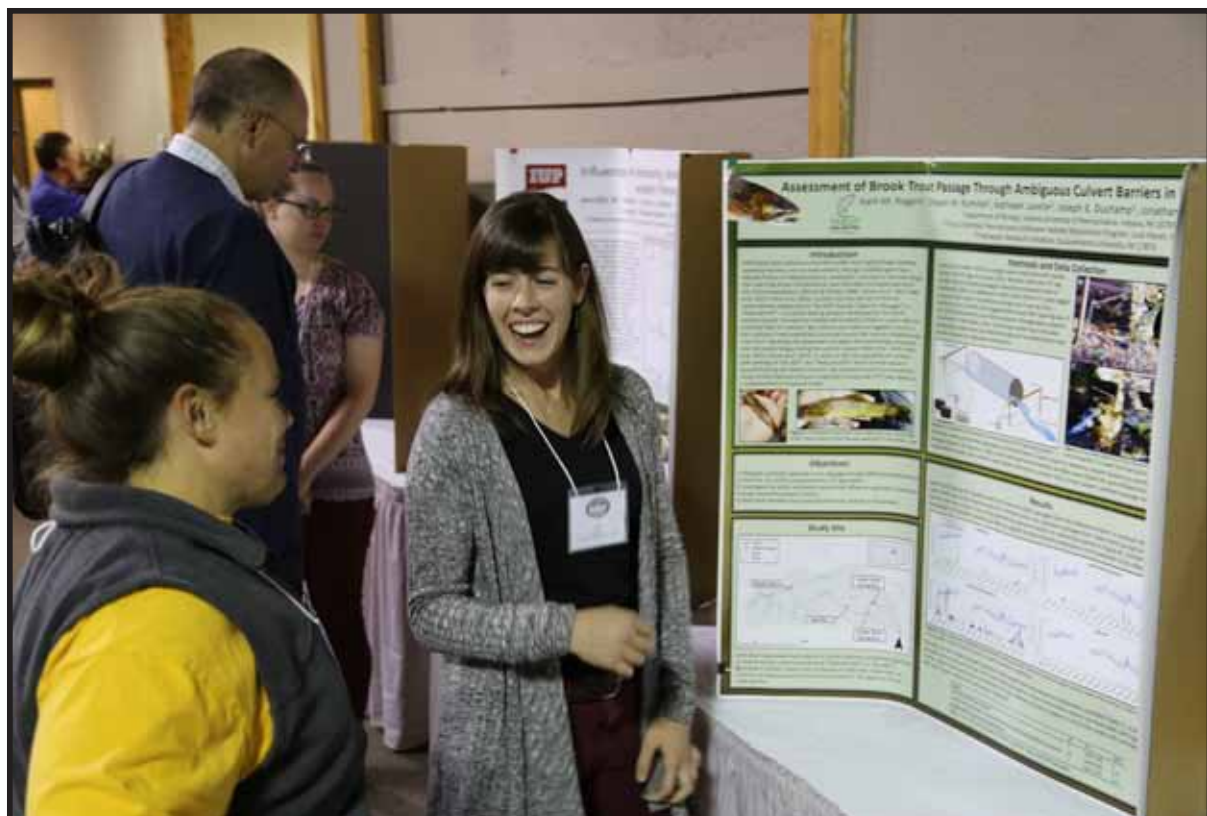
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Use of UAVs for the Inventory and Analysis of Stream Habitat

Michael P. Strager¹, Angela Hentz², Jacquelyn M. Strager³, Paul Kinder⁴, Joseph A. Kimmet⁵

¹Professor of Spatial Analysis, School of Natural Resources, mstrager@wvu.edu

²Remote Sensing Analyst, Natural Resource Analysis Center, angela.hentz@mail.wvu.edu

³Research Coordinator, Natural Resource Analysis Center, Jmstrager@mail.wvu.edu

⁴Director, Natural Resource Analysis Center, Paul.kinder@mail.wvu.edu

⁵Research Technician, Natural Resource Analysis Center, Joseph.Kimmet@mail.wvu.edu

All authors can be reached at the following address, phone and fax
Evansdale Drive, Agricultural Science Building, West Virginia University,
Morgantown, WV 26506-6108, 304-293-2941, 304-293-3742 fax

Abstract—The interaction between landscape features and receiving waters can be mapped and modeled with geospatial technologies to aid in the management of aquatic systems. Just as Geographic Information Systems (GIS) has evolved, a new spatial tool— that of capturing spatial data with an unmanned aerial vehicle (UAV) or drone has shown great potential. UAVs have data resolution and scale advantages that fit between in-person field sampling and fixed wing aircraft and imagery acquisition. UAV use for aquatic systems was applied to a high elevation native Brook Trout *Salvelinus fontinalis* stream in Central Appalachia, USA. We captured imaging of the stream using both true color and infrared to map riparian vegetation species and create normalized difference vegetation and water indices to discern water quality variations. In addition, the derived 3D point clouds allowed us to conduct “structure-from-motion” image analyses to generate landscape elevations to be used in topographic mapping and geomorphic monitoring. The advantages of this technology with proper flight planning and image processing provides great promise for improving our understanding of stream habitat.

Introduction

Riparian areas are important to support living organisms, aid in hydrological functions, and provide recreational opportunities. A riparian area is defined as the zone transitioning between a terrestrial system and an aquatic system (Pidwirny 2006). They are unique from place to place; displaying different types of soil, vegetation, wildlife, and stream characteristics, and are specifically different from upland areas in vegetation and soil. An ideal riparian system includes native vegetation and forest trees to provide shade, stability in the stream bank, woody debris, and leaf litter which are important components in the river continuum concept.

When mapping large sections of riparian areas, specifically streams, aerial imagery can be used to view these sites and make more holistic decisions regarding management. A study was conducted in the Sango Bay of Uganda using a supervised land cover classification approach -Maximum Likelihood from unmanned aerial vehicles (UAV) (Fuller 1997). The authors used 240 sample sites, classifying 14 land cover classes, and resulted in an 86% correspondence

between field and map data (Fuller 1997). Another study was conducted on the Young's Creek Watershed in the state of Indiana, where UAV imagery was also used to analyze buffer zones. The combination of remote sensing and field surveys validated the points of current buffer zones (Letsinger 2015). Both of these studies were successful in their intended use, but were heavily reliant on field data. The goal of this study was to attempt to use UAV imagery to accurately analyze the riparian area of a stream corridor of interest. We wanted to minimize the amount of field work required but still produce a viable output that can help and aid in stream and watershed management for aquatic systems. In this study, we assess UAV-enabled ultra-fine resolution riparian land cover mapping by testing the accuracy of supervised and unsupervised land cover classifications from a mosaicked set of UAV images. We also derived indices such as normalized difference vegetative index (NDVI), normalized difference water index (NDWI), a composite of true color, color infrared, and a digitized surface model (DSM) of the study site to examine the utility of the UAV products for riparian mapping and analysis.

Methods

The imagery for this study area was collected using a Phantom 3 professional UAV, equipped with two cameras, one RGB (Phantom 3 professional standard camera FC300X) and a Near Infra-Red (NIR) camera (Survey1, from MAPIR). The FC300X camera sensor has 6.317 mm x 4.738 mm of dimensions and a focal length of 3.6 mm. The NIR camera sensor has 4.911 mm x 3.683 mm of dimensions, and 5.11mm of focal length. The NIR camera presents two bands, one NIR and a blue band. The Phantom 3 professional was equipped with the RGB camera, which was mounted in a gimbal in order to remove vibrations in the camera. The NIR camera was mounted in the drone using a lateral mount, so the pictures from both cameras were acquired at the same time.

The imagery was collected September 14, 2016. The flight plan was made with Map Pilot for DJI (Drones Made Easy). We used an 80% lateral and longitudinal overlap, which allowed us to have more than 5 images of coverage for any one location. We collected 511 RGB images and 599 NIR images. The flight routes and image overlaps are shown in Figure 1.

During the imagery acquisition, we also placed and collected the coordinates of 10 targets, used as ground control during the imagery processing. The targets were positioned along both sides of the stream. Our objective was to place them in distinct aspects of the area (low and high elevations, area margins), in order to model the terrain with more accuracy. These targets are made of foam, and covered with a plastic

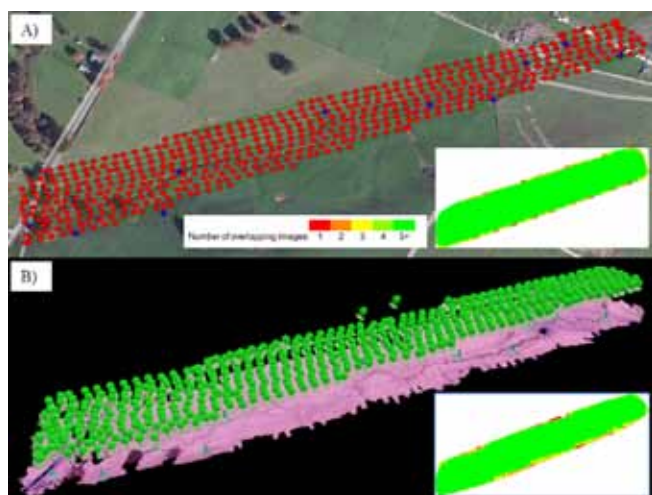


Figure 1. The points represent the centers of collected imagery. A) RGB camera; B) NIR camera.

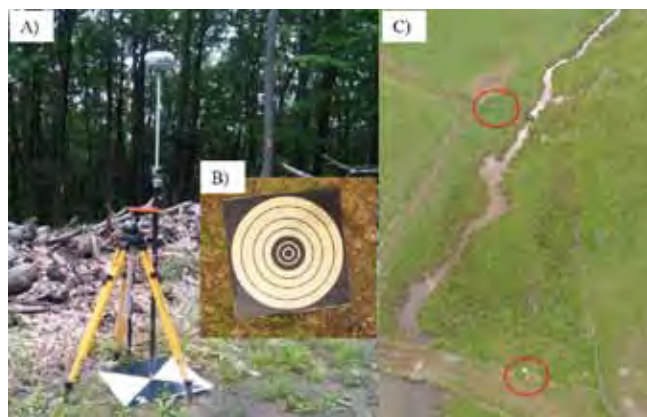


Figure 2. Ground control collected for imagery acquisition showing (A) receiver and tripod mount, (B) the ground target, and (C) an example of target positioning in the study area.

surface painted in black and yellow. The coordinates of these points were collected using an iGage X900S-OPUS GNSS static receiver, mounted at 2 m from the ground. The receiver was placed in each point for at least 16 min, and the information was sent to the Online Positioning User Service (OPUS). The OPUS corrects the positions using base stations according to the National Spatial Reference System (NSRS). The coordinate system used was NAD83 UTM zone 17N. The ground control acquisition material is presented in Figure 2.

The images were processed using Pix4D software (Pro version 2.1.58) in a Windows 64-bit computer with Intel Xeon CPU E3-1271 v3 at 3.60 GHz and 32 GB of RAM. In Pix4D there are three major steps for the orthomosaic generation.

In the initial processing, the software realizes the detection of tie point in the image, and matches them in the neighborhood images. Using this process the software can calculate the cameras positions (external orientation) as well the camera parameters (internal orientation). We applied the settings: Full Keypoints image scale, and Standard calibration (optimizing all internal and external parameters). After running this step, we included the ground control points and reoptimized the project in order to consider the ground control coordinates in the solution.

The next step involved point cloud and mesh in which a densified 3D point cloud is generated. We selected the point cloud density in Optimal (1/2 image scale).

Lastly, the DSM, Orthomosaic and index were created. In this step, we selected the options to create a DSM, and Orthomosaic as well as indices. We selected the DSM and Orthomosaic resolution at 1.5 cm, and we chose to use noise filters and a medium surface smoother. We also selected to export each band of each camera separately (with the 1.5 cm resolution). The processing was done for the two cameras separately, but the selected processing options are the same for both. During the point cloud generation process it was possible to generate 33,835,774 points (average 475.2/m³) from the RGB camera, and 45,339,878 points (average 966.28/m³) with the NIR camera. The processing from the RGB camera obtained a mean RMS error of 0.035 m, while for the NIR processing the mean RMS error was 0.012 m.

After the processing of images, the separated bands were exported and used for the index generation and band compositions. The orthomosaics, DSMs and 3D point clouds generated were exported. The products are represented in Figures 3-6.

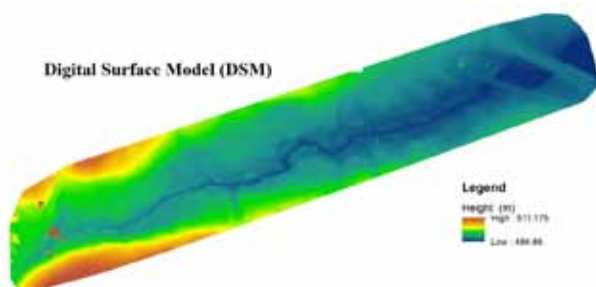


Figure 3. DSM mosaic generated using the Red Blue Green (RGB) imagery.



Figure 4. Orthomosaic generated from the RGB imagery which allows for more accurate ground measurements.



Figure 5. Orthomosaic generated from the Near Infra-Red (NIR) imagery.

In this study we also calculated two indices, the NDVI (Normalized Difference Vegetation Index) and NDWI (Normalized Difference Water Index), based in the following formulas:

$$NDVI = \frac{\rho_{NIR} - \rho_{Red}}{\rho_{NIR} + \rho_{Red}}$$

$$NDWI = \frac{\rho_{Green} - \rho_{NIR}}{\rho_{Green} + \rho_{Red}}$$

Where: ρ_{NIR} , ρ_{Red} and ρ_{Green} are near-infrared band, red band, and green band reflectance, respectively.

To calculate the index, we used bands from two cameras, green and red from the RGB camera, and NIR from the NIR camera. The index calculations were made in ArcMap 10.4, using the Raster calculator tools.

In the same way, we created a composition using 4 bands (Near InfraRed, Red, Green and Blue), which was used for the classification process. Both cameras have a Blue band, and in this study we decided to use the blue band derived from the RGB camera, since it has a better resolution. The NDVI and NDWI index, as well the composition in false color are presented in Figures 7-9.

Results

Using the NDVI proved useful to understand the parts of the riparian area with vegetation since healthy vegetation generally has a higher leaf water content compared to other areas (Maarel 2009). When using these data as a layer within GIS for the study site, it helped us delineate riparian vegetation from outlying vegetation and from the actual water in the tributary.

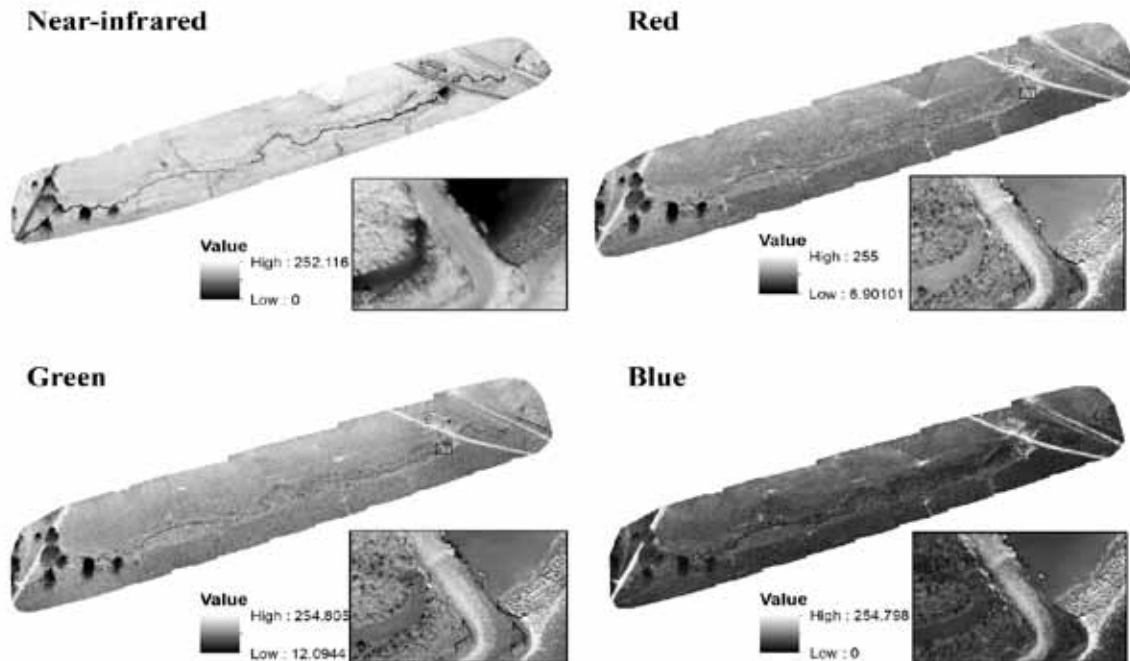


Figure 6. The individual imagery bands demonstrate the contrast and importance of information extractable from different sources.

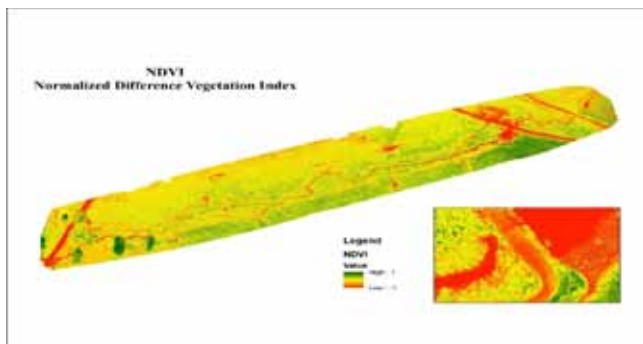


Figure 7. The NDVI used to indicate growing healthy vegetation.



Figure 9. Darker red areas showing healthy growing vegetation.

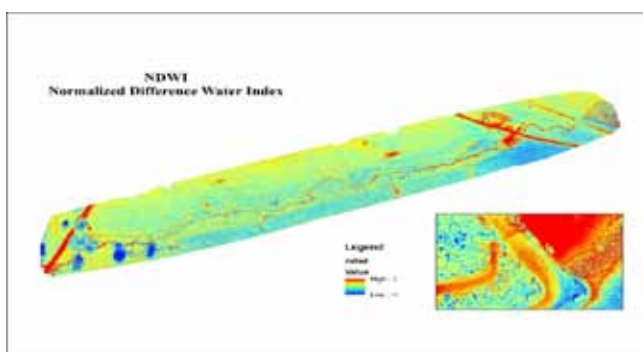


Figure 8. NDWI complements the NDVI by showing vegetation liquid water.

The UAV imagery was also helpful to produce NDWI data to differentiate between the water content in vegetation and the water content in actual bodies of water. Using this as a layer in GIS, in correlation with NDVI helped to define the tributary as compared to the riparian vegetation.

The composite imagery created for this study site made use of the classification tools and a composite image called NRGBD. It consisted of five bands that include NIR or true color, Red, Green, Blue, and a DSM. We processed these images with a photogrammetric software, extracted the bands, and

combined the bands (R, G, B and NIR). In this image the red indicates healthy, live vegetation while the blue and green indicate water and other hydrated areas. The DSM is a digitized surface model that shows us the elevation and surface changes in our study site. The relationship among these different bands within this composite image was useful in identifying patterns and correlations between a specific layer and other layers, as well as to the landscape itself. The ability to see pool and riffle extents gave insight into the structural habitat.

Discussion

While these tools were mostly successful, there are many challenges with these classification tools. The human eye does not always have a definitive interpretation of these ortho-photos so choosing feature samples can be skewed. Also, natural shadows in the landscape and stream cannot be viewed as clearly and can be misclassified. In some of the classifications, debris, fence, roads, and tree crowns were classified incorrectly. Although the Maximum Likelihood classification tool worked well for this study site, a larger number of feature class polygons could have been added to delineate these features the best.

Aside from the challenges that appear using UAV imagery to classify land cover and riparian areas, and the uncertainties in the classification tools, the images produced for this study site gave us positive insight on specific relationships. Looking at the contour of the elevation in the DSM layer, at the specified clip of stream, closely relates to the difference in land cover classes output by the ISO Cluster function. Where the riparian area is obviously disconnected from the changing landscape, the elevation also lowers into the riparian area, and the change in visual value in the ISO Cluster maps of NRGD, NDWI, and NDWI show this as well. This relationship is just one of the many important keys to showing us what these riparian areas consist of at all levels and where buffer zones should exist depending on the specific site.

Acknowledgements

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findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of the National Science Foundation.

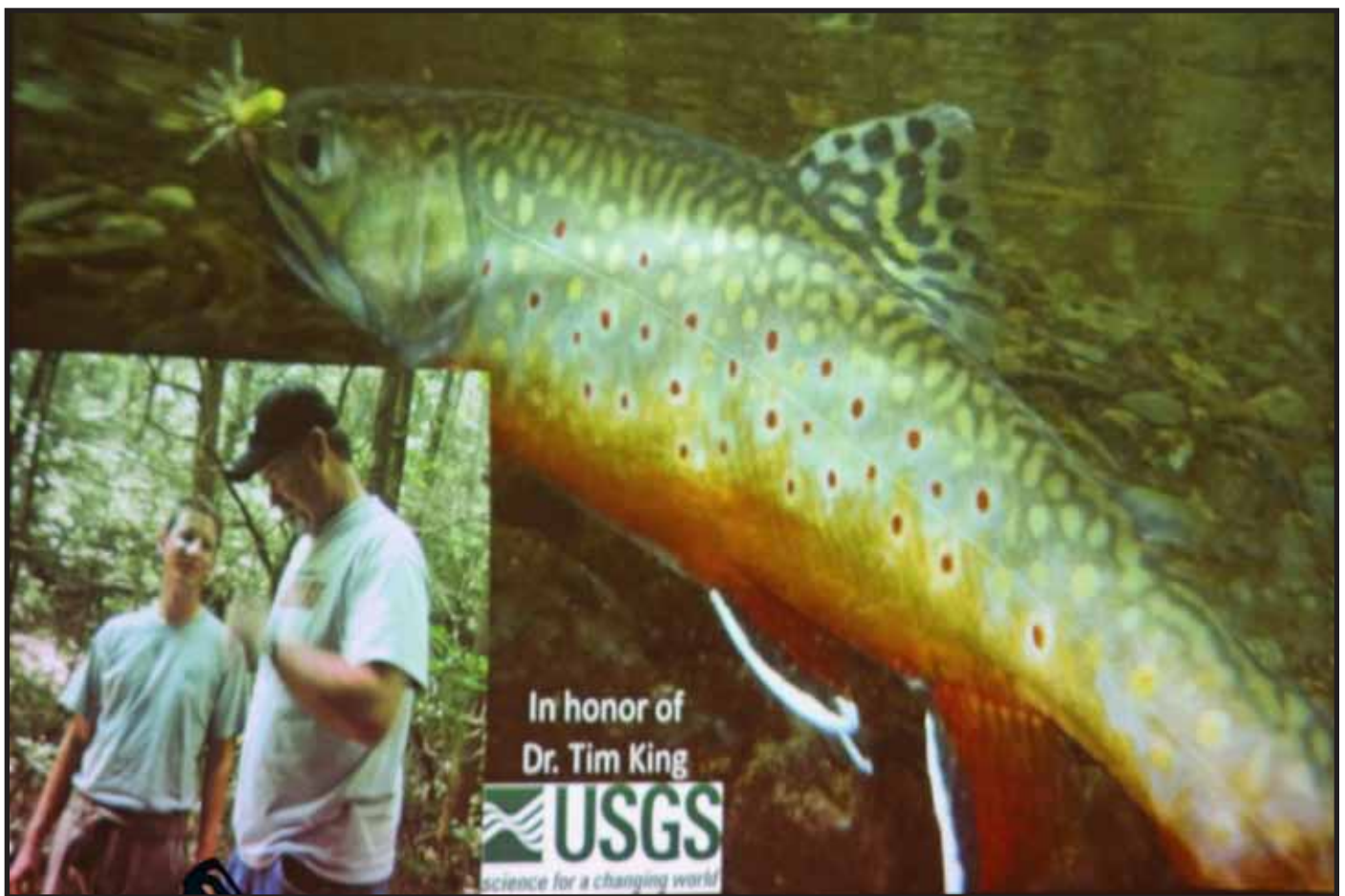
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Session 3

Conservation Genetics and the Genomics of Coldwater Fishes: A Tribute to Dr. Tim King



In Memoriam: Dr. Timothy L. King (1958-2016)

S.P. Faulkner and D.C. Kazyak

USGS Leetown Science Center, 11649 Kearneysville Rd.,
Leetown, WV 25430

Tim King was an internationally recognized expert in the field of conservation genetics and genomics, using molecular techniques to understand the evolutionary basis of genetic variation and adaptation. His research contributions provided the basis for identifying unique species and populations and have guided state and federal conservation programs of threatened and endangered species. He published over 100 research papers on the genetics and genomics of salmon, brook trout, sturgeon, turtles, horseshoe crab, deep-sea coral, Key Largo woodrat, Preble's meadow jumping mouse, black bear, and invasive species.

Tim began his career as a Conservation Biologist with the Texas Parks and Wildlife Department in Palacios, Texas. After receiving his Ph.D. from the University of North Texas, he joined the U.S. Department of Interior, Leetown Science Center, in Kearneysville, WV in 1993. The Center became part of the U.S. Geological Survey in 1994 where Tim led the development of the genetics program until his death. He worked tirelessly to leverage new technologies to enhance the conservation and management of imperiled organisms. In addition to his prolific research productivity, he was often asked to serve as an expert Federal government scientist on threatened and endangered species issues. Tim was appointed as the North American expert on the Atlantic salmon to assess the potential impacts of genetically engineered aquaculture salmon on wild populations. He was awarded the U.S. Geological Survey Superior Service Award in 2002 for his significant contributions to understanding the biology of Atlantic salmon that was invaluable to Federal and State managers in their efforts to recover this important species.

Tim began working on brook trout in the early 2000s, developing a suite of microsatellite markers that have proven to be invaluable molecular tools for studying this species. Under Tim's supervision, staff at the Leetown Science Center genotyped over 21,000 individual brook trout from over 800 locations across their native range, offering unprecedented insight into the structure and function of wild populations. Tim was especially



interested in the conservation of brook trout in the southern Appalachians, where hundreds of small, isolated populations face unique challenges. He delineated biologically-appropriate management units, identified metapopulations, and characterized the extent of hatchery introgression in wild populations. He worked extensively with resource managers, acting as a tireless advocate for conservation genetics and identifying appropriate source stocks for translocation efforts. In recent years, Tim began applying next-generation genomics techniques to brook trout, working towards an understanding of adaptive differences among wild populations. Tim's insights continue to help guide the restoration and management of wild brook trout populations.

In addition to his scientific contributions, Tim was always in great demand as a collaborator, generous with his time and expertise, and a tremendous ambassador for science who readily shared his enthusiasm with many students and colleagues. He gave over 200 presentations during his career ranging from meetings of local stream associations to university classrooms to invited presentations at international scientific meetings of the world's most knowledgeable scientists. He mentored 19 graduate students and numerous post-doctoral scientists and was an adjunct faculty member at Pennsylvania State University, West Virginia University, University of Toledo, Frostburg State University, and Montclair State University. His visionary leadership, keen analytical mind, quick wit, kindness, and friendship was taken from us much too soon and will be sorely missed.

Genetic Structuring of Brook Trout in Exurban Riverscapes: Is it all About the Dam Barriers?

Lucas R. Nathan¹, Amy B. Welsh², and Jason C. Vokoun¹

¹University of Connecticut, Wildlife and Fisheries Conservation Center, Department of Natural Resources and the Environment

²West Virginia University, Division of Forestry and Natural Resources

Extended Abstract

Stream and river networks are often fragmented by physical barriers, which can restrict fish passage and disrupt ecosystem functioning. Brook Trout *Salvelinus fontinalis*, like most wild trout species, are threatened by habitat fragmentation and losses due to anthropogenic influences. Contemporary populations across much of their native range are limited to small headwater streams due to increasing main-stem river temperatures and the presence of physical barriers reduce dispersal among and within streams. Limiting dispersal of individuals can reduce gene flow between populations and disrupt metapopulation dynamics, which in turn can decrease population adaptability and viability. Removing barriers, while effective as a conservation strategy, can be difficult in exurban landscapes where small low-head dams and road crossing culverts exist in high densities throughout stream networks. Prioritization strategies to date have focused on a “miles upstream” approach, where barriers are selected for removal based on the amount of river habitat reconnected. This strategy, however, fails to recognize heterogeneity in habitat and does not account for cryptic barriers, such as water quality, that may also influence fish movement. Evaluations of Brook Trout dispersal at the watershed scale have been limited to date due to the logistical challenges of conducting capture-mark-recapture studies at larger spatial scales. Landscape genetics analyses, aimed at exploring relationships between landscapes and microevolutionary processes, have gained popularity over the last 15 years, yet are underutilized in studies of riverscape systems. Although the influence of watershed land use on Brook Trout occupancy and development has been documented, such influences have not been included in evaluations of Brook Trout dispersal and population connectivity. The objectives of this research were to (1) estimate genetic differentiation of Brook Trout among headwater streams and (2) use a riverscape genetics approach to identify features limiting gene flow. The findings

of this research will be used to improve conservation and management strategies by identifying areas of conservation opportunity, both in terms of protection of valuable dispersal habitats and restoration efforts to promote population connectivity and increases in long term viability.

We collected Brook Trout fin clips for genetic analyses at 81 sites in two watersheds in North Central Connecticut, each with a surface area of approximately 400 mi². We genotyped all individuals using eight microsatellite loci and estimated pairwise genetic differentiation (F_{ST}). Using a two-step cost-distance optimization approach within a causal modeling framework, we identified potential features limiting gene flow among headwater streams in each watershed. Using ArcGIS Network analyst and data from the State of Connecticut’s Department of Energy and Environmental Protection website (www.ct.gov/deep), we identified barriers (number of dams, Presence or Absence of dams, number of road crossings) as well as stream distance between sites. From the SHEDS database (www.ecosheds.org), we quantified nine landscape variables (slope, elevation, % development, % impervious cover, % forest, % wetlands, % agriculture, % tree canopy, and watershed area) and two riverscape variables (% coarse surficial geology and maximum annual stream temperature) at five spatial levels (50 ft-riparian [i.e. 50 ft per side], 100 ft-riparian, 200 ft-riparian, local stream reach catchment, and total upstream watershed). We then used a two-step optimization approach to identify features correlated to genetic distances. First, variables were converted to cost-distance values using the equation,

$$CD = (\text{Value}/\text{max Value})^x * \text{maxCost}$$

where *value* was the raw variable value and *x* was the rate at which the cost-distance value approached the *maxCost* held at a constant of 100 during the first optimization procedure. The relationship between cost distances, summed between each pairwise site,

and genetic distances (F_{ST}) was evaluated with simple and partial Mantel tests with significance based on 999 permutations. During the second step in the optimization process, landscape variables that had a significant correlation with genetic distances while controlling for the effects of river distance and barriers were combined using 1,000 combinations of *maxCost* while holding the *x* parameter from step one constant. Multivariate surfaces were evaluated in the same way as step one, with simple and partial Mantel tests using pairwise genetic distances and summed cost-distances.

Pairwise differentiation between sites was highly variable, ranging from 0.01 to 0.32 with a mean F_{ST} of 0.12. While some had lower levels of differentiation with neighboring sites, consistent with sub- or meta-population structuring, others were functioning as isolated populations with higher levels of differentiation. Two variables, percent upstream development and percent upstream impervious cover, were significantly correlated with genetic distances in both watersheds after controlling for the effects of distance and barriers. A third variable, 50-ft and 100-ft riparian slope, was significant for the western watershed, but not the eastern. Stream distance and dams were also significantly related to genetic distances, however with lower correlations compared to the landscape variables (e.g. eastern watershed $\text{Mantelr}_{\% \text{upstrmdev}, \text{dams}, \text{p/a dam}} = 0.47, 0.18, \text{ and } 0.15$). Due to the high correlation between percent development and percent impervious cover ($r = 0.94$) and based on relative Mantelr correlation statistics, only 100-ft riparian slope and percent upstream impervious land

cover were included in the multivariate optimization. The combination of the two cost-distance variables did not produce a higher correlation to genetic distance than percent upstream development alone ($\text{Mantelr}_{\text{eastern, western}} = 0.47, 0.46$). The cost-distance reclassification function that resulted in the highest correlation to genetic distances was an exponential relationship, where percent upstream development of greater than 20% was correlated with increases in genetic distances. These results suggest that watershed development decreases gene flow in headwater Brook Trout populations.

We documented the combined influence of physical barriers and landscape-level variables on the genetic structuring of Brook Trout. The influence of land use types on Brook Trout occupancy in headwater streams, particularly developed and impervious land cover, has been well documented. From a connectivity perspective, however, dams and road crossings have been the primary focus for Brook Trout conservation. Our results suggest that a combination of physical barriers and landscape features influence genetic connectivity between headwater populations, and thus both should be incorporated into conservation planning and prioritization. Prioritizing barrier removals using only a “miles upstream” approach is unlikely to result in the most effective management strategies if landscape level influences are not accounted for. Moving forward we will be developing decision support tools using this landscape-based approach to help managers identify specific areas across the state and region that should be targeted for conservation actions.

Understanding the Genetic Characteristics of Wild Brook Trout Populations in North Carolina Thanks to the Guidance of Dr. Tim King

D.C. Kazyak¹, B.A. Lubinski¹, J.M. Rash², and T.L. King¹

¹USGS Leetown Science Center, 11649 Leetown Rd, Kearneysville, WV 25430

²Division of Inland Fisheries, North Carolina Wildlife Resources Commission, 645 Fish Hatchery Road, Marion, NC 28752, USA

Abstract—We genotyped 7,588 brook trout representing 406 collections from across the State of North Carolina (Figure 1) at 12 microsatellite loci (King et al. 2012). The vast majority of collections appeared to represent single populations, based on general conformance to Hardy-Weinberg equilibrium and limited evidence for linkage-disequilibrium. Allelic diversity was low to moderate relative to Brook Trout *Salvelinus fontinalis* populations endemic to higher latitudes. Effective population sizes varied widely among populations, but were often very small and indicate that many populations are at risk of losing diversity through genetic drift. Remarkable levels of genetic differentiation exist among populations, which suggests that little, if any, gene flow occurs among most populations. Analysis of molecular variance (AMOVA) revealed that a substantial portion of the observed genetic variation was attributed to differences among patches (44.8%), and there was some variation (11.2%) even among collections within a single patch. These results, taken in conjunction with high levels of genetic differentiation among populations, suggest that the fundamental unit of management for Brook Trout should be the population. Interestingly, despite extensive stocking across the state, the vast majority of wild populations show limited evidence of introgression by northern origin hatchery strains. These results represent a valuable baseline for management and restoration efforts, and can be used to (a) select suitable donor streams for translocation efforts, (b) identify streams with low effective population sizes that may be vulnerable to extirpation, and (c) target stocking efforts into watersheds where extensive introgression has already occurred. All data associated with this manuscript has been publicly released (Kazyak et al. 2017).

Specific Analyses

Patterns of Diversity

Descriptive statistics for each collection were generated using GenAlEx 6.502 (Peakall and Smouse 2006, 2012). The percentage of polymorphic loci, allelic richness (N_a) and observed heterozygosity (H_o) were calculated for each population. In general, most of the 12 microsatellite loci were polymorphic in the majority of populations (Figure 2). Allelic richness was typically low to moderate relative (mean 3.13 alleles/locus; range: 1.00-6.42) to brook trout populations endemic to higher latitudes, but varied considerably among collections (Figure 1 and 3). Observed heterozygosity also varied among populations taken from across the state (mean: 0.42; range: 0.00-0.76; Figures 4-5).

Do Collections Represent Populations?

The vast majority of the collections appeared to represent single, well-mixed populations, as supported by their conformance to Hardy-Weinberg equilibrium

(tested using Genepop v. 4.3) and limited evidence for linkage-disequilibrium (Figures 6-7). There is no evidence for the widespread occurrence of multiple populations occurring in sympatry within a single stream reach.

Effective Population Size

Single sample estimates of effective population size (N_e) based on linkage disequilibrium were produced using NeEstimator v2 (Do et al. 2014), using a rare allele cutoff of 0.02 and jackknifed confidence intervals. We used N_e rather than N_b because our collections included samples with mixed generations, rather than single cohorts. Effective population sizes varied widely among populations, but were often very small (134 collections had an estimated $N_e < 10$). Although these estimates should be interpreted with caution in those collections consisting of a small number of individuals, it is clear that many populations (provided in the excel spreadsheet), are at risk of losing diversity through genetic drift.

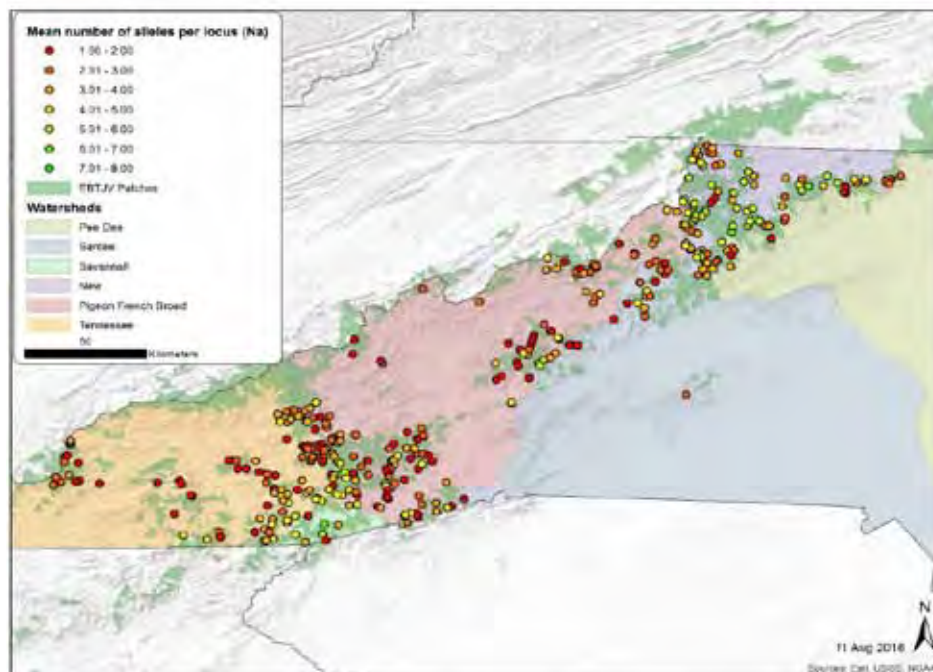


Figure 1. We genotyped 7,588 individual brook trout representing 406 collections across six major watersheds in western North Carolina. Geographic variation in allelic diversity (N_a) is represented by the color of each point

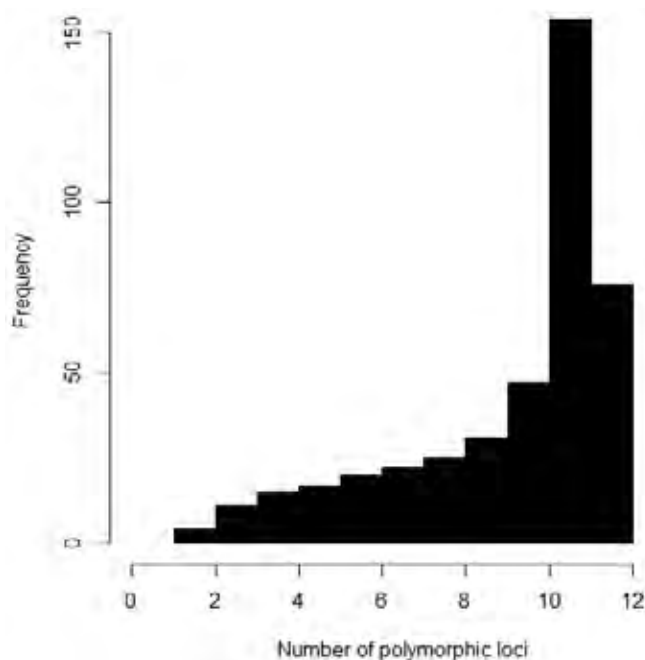


Figure 2. Histogram of the number of polymorphic loci observed across wild brook trout collections from North Carolina.

Differentiation Among Populations

We examined genetic differentiation among populations using pairwise F'_{ST} values (Meirmans and Hedrick 2011) calculated with the *diversity* package (Keenan et al. 2013) in R (R Core Team 2015). High levels of genetic differentiation (mean: 0.72; range: 0.00-0.99) were found among nearly all collections (Figure 8). This suggests that little to no gene flow is occurring among most populations, and that hatchery stocking has not had a strong, widespread homogenizing influence on the genetics of native brook trout.

Distribution of Genetic Variation Across North Carolina

In order to examine the geographic structure of genetic variation, we used a hierarchical Analysis of Molecular Variance (AMOVA), implemented with the *pegas* package (Paradis 2010) within R (R Core Team 2015). Three hierarchical levels were considered: collection, patch, and ecological drainage unit. The patches were developed by the Eastern Brook Trout

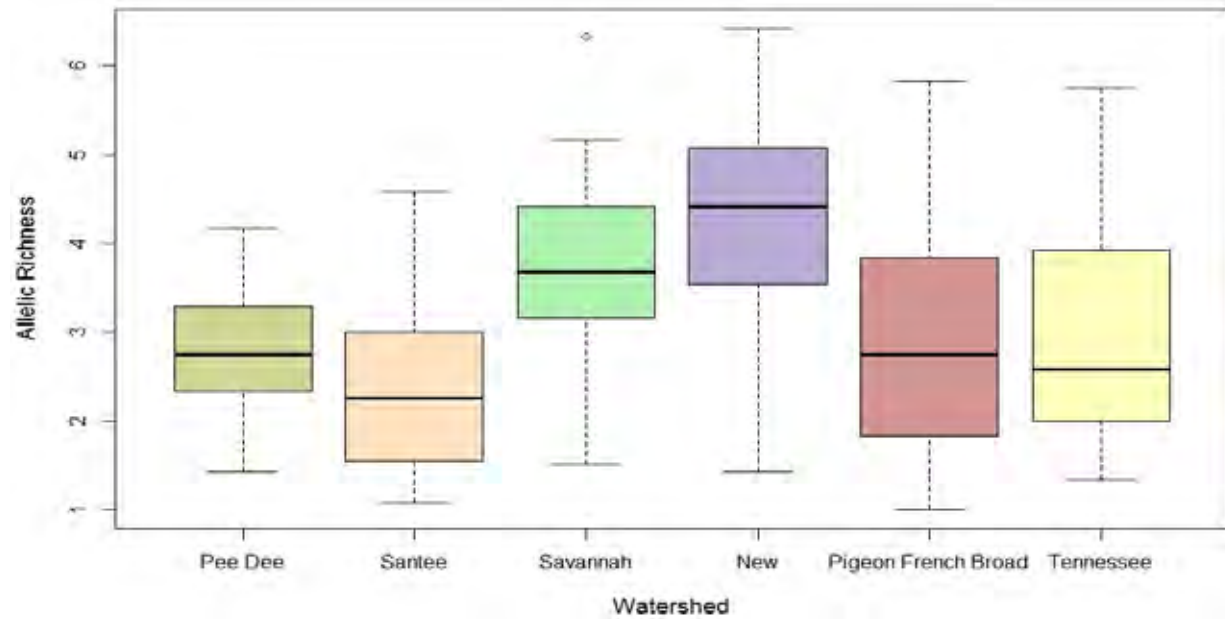


Figure 3. Variation in allelic richness (N_a) among brook trout collections taken from six major watersheds across North Carolina.

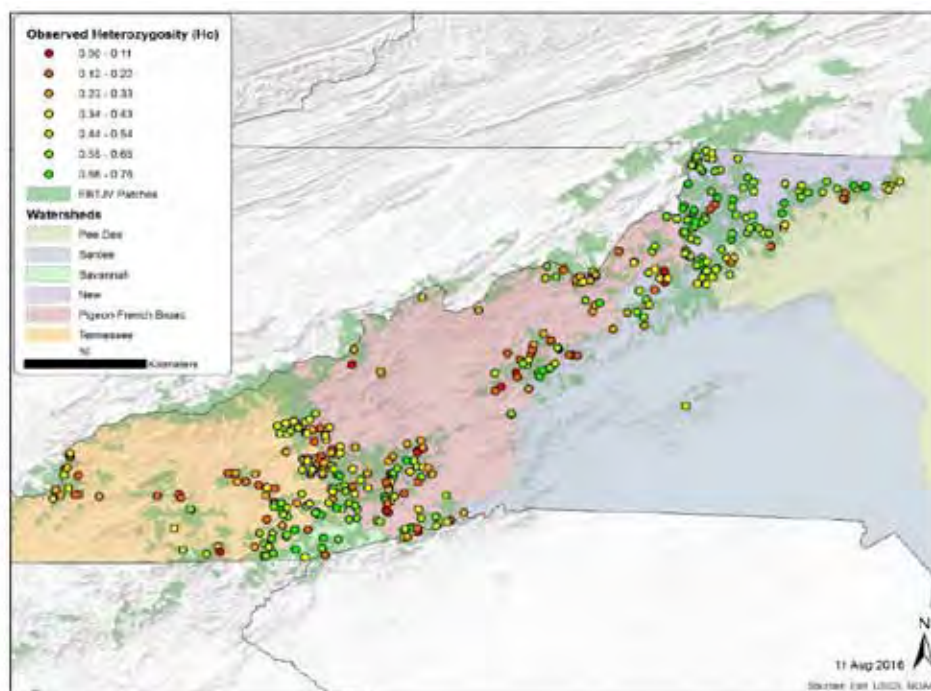


Figure 4. Geographic variation in observed heterozygosity (H_o) in wild brook trout populations from western North Carolina.

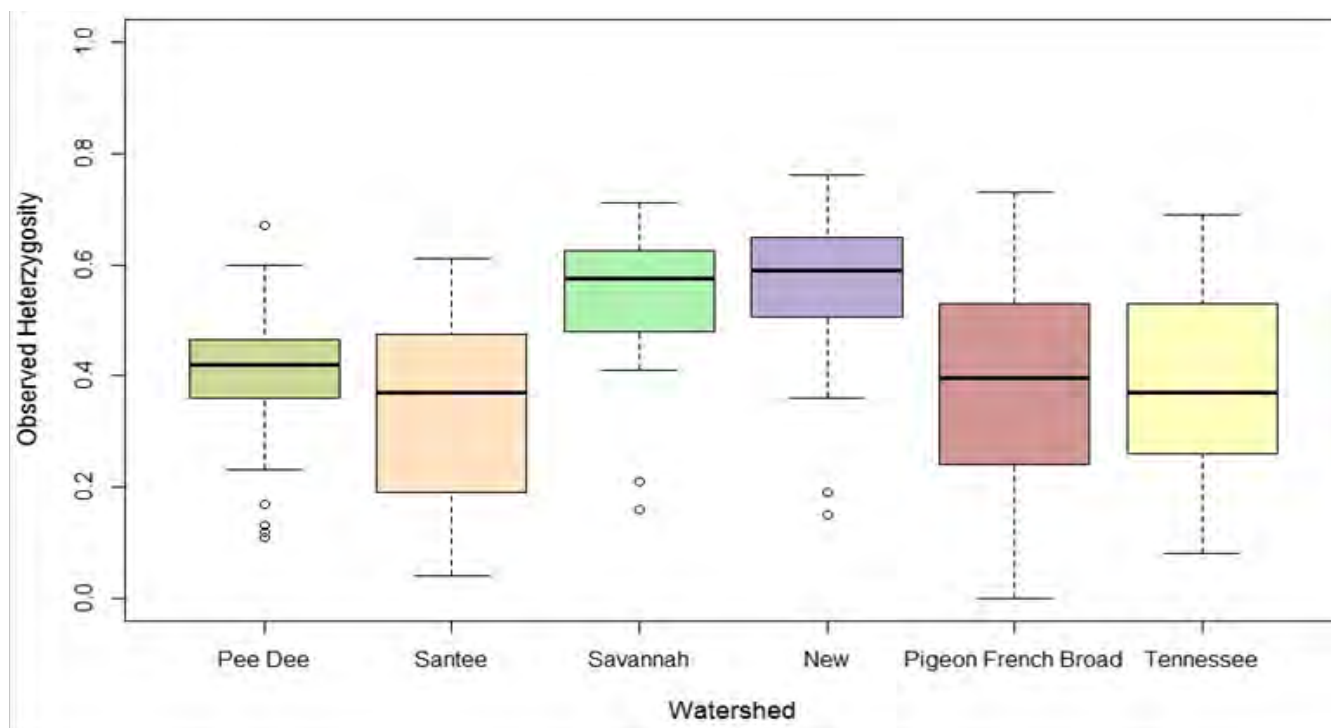


Figure 5. Variation in observed heterozygosity among brook trout collections taken from six major watersheds across North Carolina.

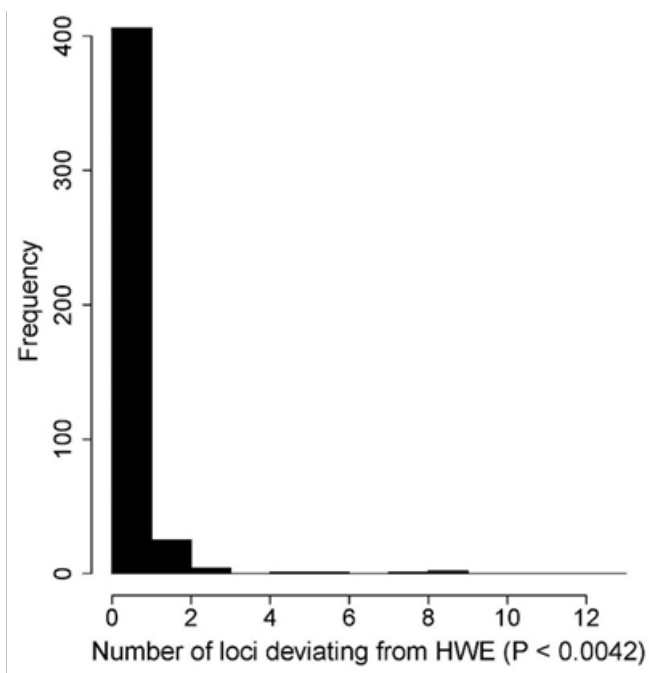


Figure 6. Histogram showing the number of loci with statistically significant ($P < 0.0042$) deviations from Hardy-Weinberg equilibrium (HWE) across wild brook trout collections from North Carolina.

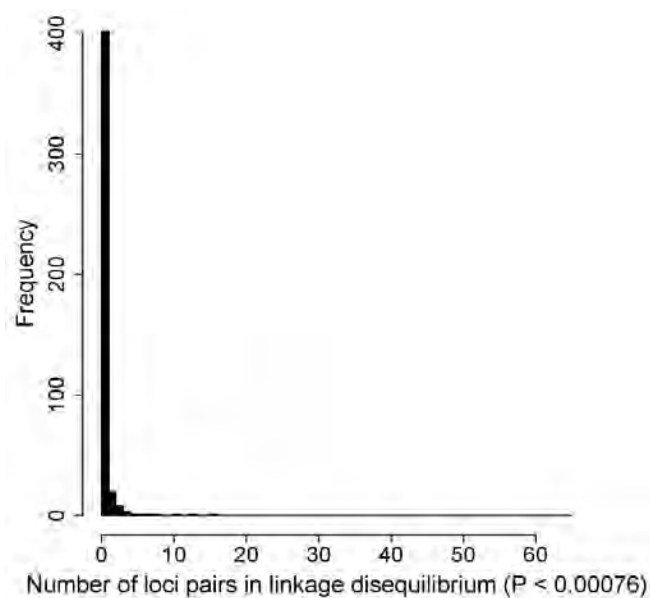


Figure 7. Histogram showing the number of pairs of loci with statistically significant ($P < 0.00076$) linkage disequilibrium across wild brook trout collections from North Carolina.

Table 1. Hierarchical analysis of molecular variance (AMOVA) for 406 collections across North Carolina.

	SSD	Variance Explained
<i>Among EDUs</i>	1250158	8.0%
<i>Among patches within EDUs</i>	6973954	44.8%
<i>Among collections within patches</i>	1751710	11.2%
<i>Among individuals within collections</i>	5605650	36.0%
Total	15581471	

Joint Venture, and are intended to represent contiguous stream habitats that support Brook Trout. Conversely, ecological drainage units (EDUs; Higgins et al. 2005) cover much larger areas and are expected to encompass many populations. A small proportion of the genetics collections was not located within a patch. For the purposes of this analysis, those collections were not considered in the AMOVA or assignment tests.

We found that the majority of observed genetic variation occurred at small spatial scales (Table 1).

Nearly half of all variation was attributed to differences among patches (44.8%), and there was some variation (11.2%) even among collections within a single patch. These results, taken in conjunction with high levels of genetic differentiation among populations, suggest that the fundamental unit of management for Brook Trout should be the population. In the absence of high resolution information to delineate populations, patches should provide a tractable surrogate for management and conservation.

Genetic Assignment Testing

To further assess the uniqueness of each collection, we assigned each individual to a specific source collection. Assignment testing was conducted using GeneClass2 (Piry et al. 2004) based on the Bayesian approach of Rannala and Mountain (1997). Classification efficiencies to the correct collection, patch, and EDU were calculated and expressed as a percentage. Nearly all individuals were assigned to their EDU of origin (99.1%). Remarkably, the vast majority (94.0%) of Brook Trout were assigned to the correct patch, and many even to the correct collection (86.0%). The assignment testing results highlight the fine-scale genetic structure prevalent in Brook Trout populations across North Carolina. Additionally, we conducted similar assignment testing using the individuals presented in this report along with >10,000 fish in >400 collections from elsewhere in the range. Despite the additional opportunities for classification error, assignment success was still exceptionally high for populations from North Carolina, further reinforcing that wild populations are individually unique.

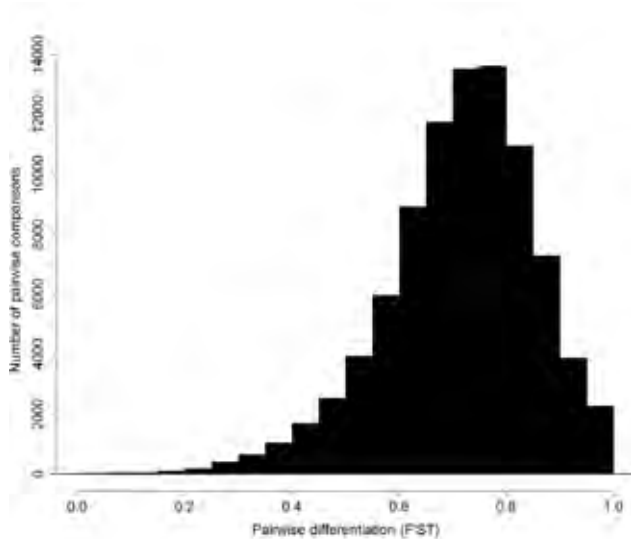


Figure 8. Histogram of pairwise F'_{ST} values among 406 wild brook trout collections in North Carolina.

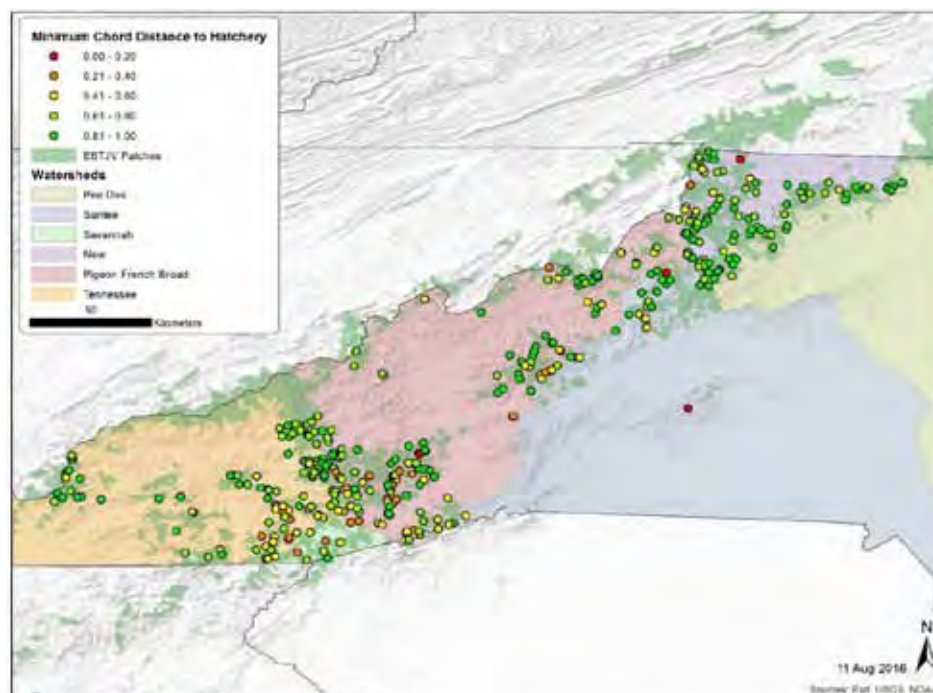


Figure 9. Geographic variation in genetic distance to northern origin hatchery reference collections across brook trout populations in western North Carolina.

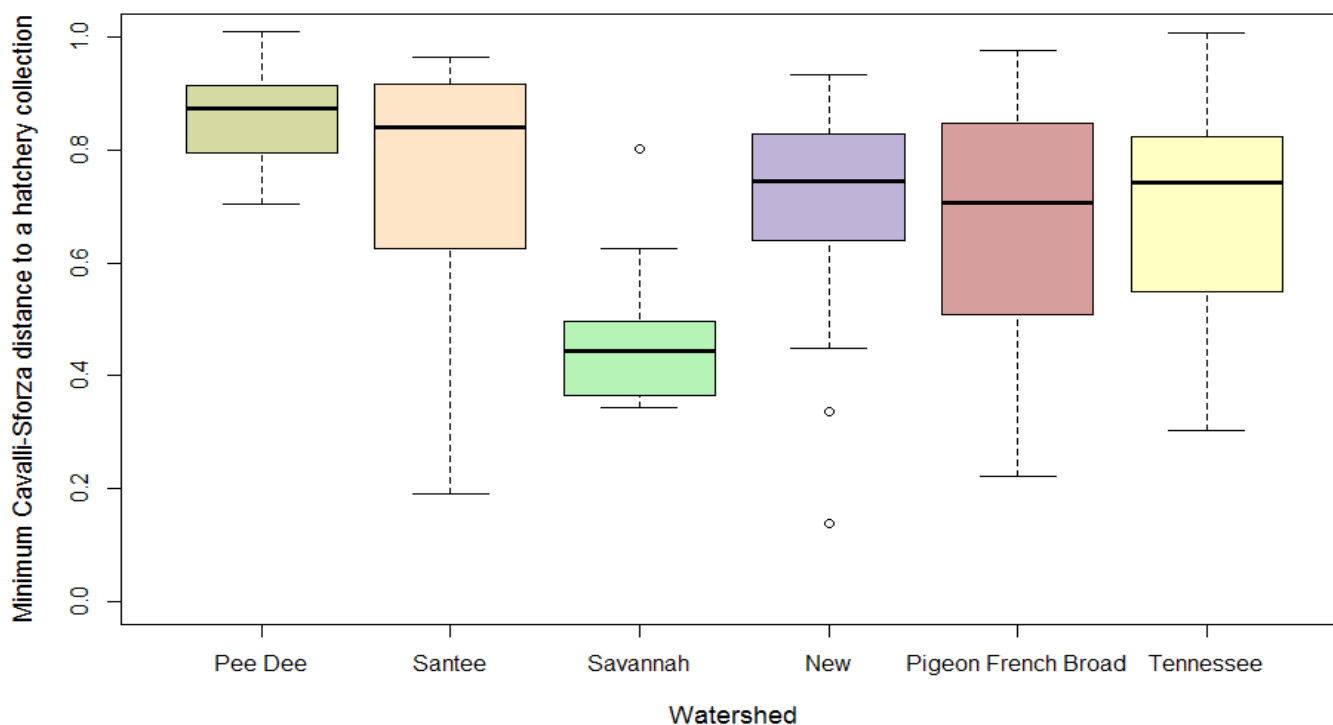


Figure 10. Variation in the minimum genetic distance to a hatchery reference collection among brook trout collections taken from six major watersheds across North Carolina.

Hatchery Introgression

In an effort to characterize the extent of hatchery introgression within wild populations, we genotyped 18 collections of hatchery brook trout provided from within North Carolina and throughout the eastern United States. We calculated Cavalli-Sforza chord distances (Cavalli-Sforza and Edwards 1967) among each hatchery collection and each wild collection using the hierfstat package (Goudet and Jombart 2015) in R (R Core Team 2015). Most collections were genetically distant ($D_C > 0.5$; Figure 9) from the hatchery strains, suggesting that these collections still retain much of their endemic character and have not been strongly influenced by stocking. However, a small number of collections were genetically similar to hatchery strains. At these sites, brook trout populations may have been founded by hatchery progeny or high levels of introgression are likely to have occurred. Interestingly, we found significant differences in the apparent extent of hatchery introgression among major watersheds (Figure 10), with the Pee Dee River exhibiting little to no stocking influence and the Savannah River being the most strongly impacted.

Acknowledgements

Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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Metapopulation Dynamics, Landscape Genetics, and the Effect of Isolation Upon Neutral Genetic and Phenotypic Variation in Southern Appalachian Brook Trout

T. Casey Weathers¹, John Carlson¹, Dave Kazyak², Matt Kulp³, David Walter⁴, Ephraim Hanks⁵, Jesse Lasky⁶, Steve Moore⁷, and Jake Rash⁸

¹Department of Ecosystem Science and Management, The Pennsylvania State University, 321&323 Forest Resources Building University Park, PA 16802;

²US Geological Survey, Biological Resources Division, Leetown Science Center, Aquatic Ecology Laboratory, 11649 Leetown Road, Kearneysville, WV 25430;

³National Park Service, Resource Management and Science Division, Great Smoky Mountains National Park, 107 Park Headquarters Road, Gatlinburg, TN, 37738;

⁴Department of Ecosystem Science and Management, The Pennsylvania State University, 403 Forest Resources Building University Park, PA 16802;

⁵Department of Statistics, The Pennsylvania State University, Millennium Science Complex W-250 University Park, PA 16802;

⁶Department of Biology Eberly College of Science, The Pennsylvania State University, 408 Life Sciences University Park, PA 16802;

⁷Retired-National Park Service, Resource Management and Science Division, Great Smoky Mountains National Park, 107 Park Headquarters Road, Gatlinburg, TN, 37738;

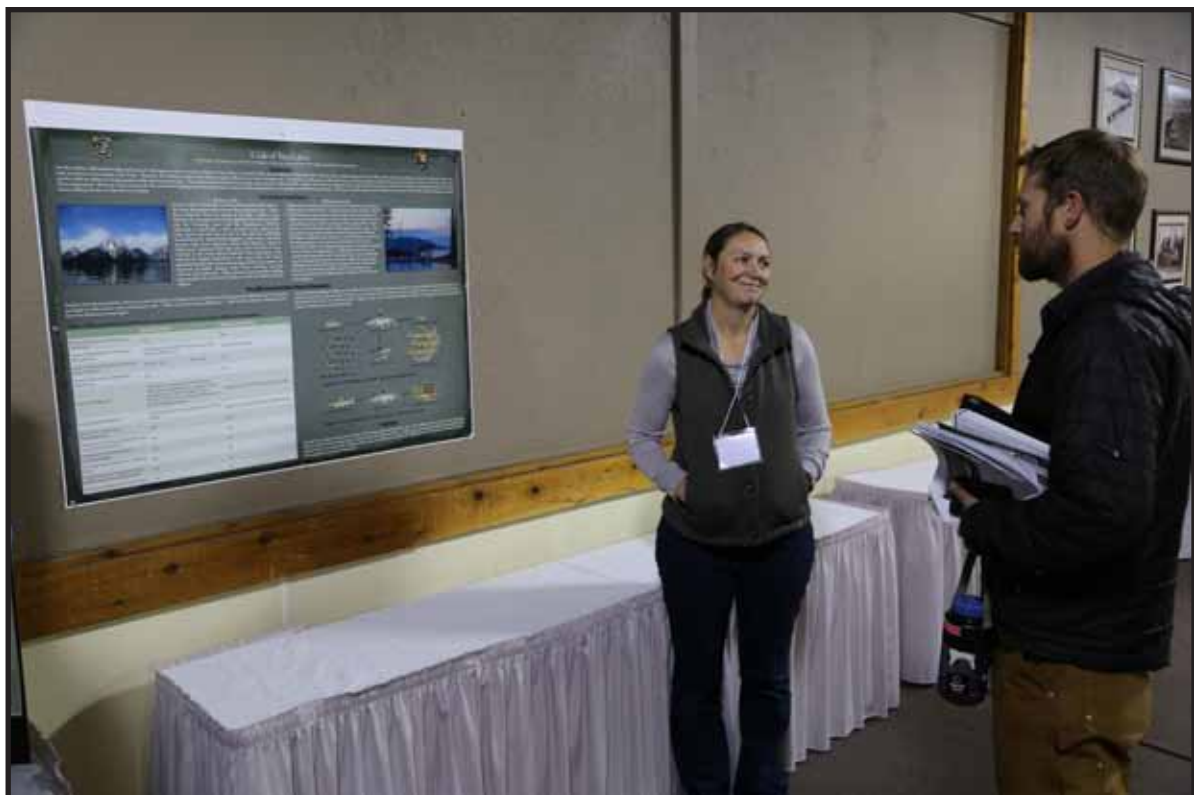
⁸North Carolina Wildlife Resources Commission Inland Fisheries Division, 645 Fish Hatchery Road, Marion, NC 28752

Abstract—Limited habitat connectedness among barrier-laden headwater streams is thought to reduce gene flow among fragmented Brook Trout *Salvelinus fontinalis* populations. These isolated populations typically have low effective population sizes and are especially vulnerable to genetic drift. Here, we present a synopsis of a doctoral research project that sought to identify eco-spatial barriers associated with genetic discontinuity. Initial assessments of neutral genetic diversity and differentiation (F_{ST}) were used to estimate effective migration rates and identify signatures of metapopulation dynamics. We then implemented a landscape genetics approach using Mantel tests followed by a series of multiple regression on distance matrices (MRM) to examine how unique landscape features potentially effect each fishery. Next, we examined how isolation and genetic drift relate to phenotypic variation within and among populations. Viewed collectively, these results offer insight into what landscape features are associated with sustained gene flow and metapopulation structure, and help us understand how habitat fragmentation coupled with genetic drift has contributed to population differentiation, reduced levels of genetic diversity, and observed phenotypic variation. This information provides managers insight into landscape scale population structuring and identifies the habitat variables that may be feasible to alter to restore gene flow among fragmented populations. In the future we hope to apply next generation sequencing techniques to gain more insight into population-specific adaptations and delineate potential conservation units.

Session 4

Population Dynamics and the Ecology of Wild Trout





Lessons from Long-term Studies of Trout Population Dynamics

Gary Grossman

Warnell School of Forestry & Natural Resources, University of Georgia, Athens, GA 30602

Extended Abstract

My colleagues and I have analyzed multiple long term quantitative data sets for Brook Trout *Salvelinus fontinalis*, Rainbow Trout *Oncorhynchus mykiss*, and Brown Trout *Salmo trutta* ranging from 10 to 50+ years and found evidence of density dependence in the per capita rate of increase and growth for all studies. Findings of density-dependence are important from both a basic and applied perspective, because density-dependence allows populations to respond in a compensatory manner when mortality is catastrophic or overfishing occurs. Our results stimulated an interest in the frequency of density-dependent effects in nine species of salmonids, so we began a survey of the published literature. Specifically, we evaluated the relative importance of density-independent (typically flow and temperature variation) and density-dependent factors (abundance) on demographic parameters including: abundance/density, recruitment, fecundity, survivorship and growth. At present our data base includes 90+ studies with more to come. We examined the studies for the importance of the following effects: no density-dependence or density-independence, density-independence, and density-dependence. The vast majority of studies displayed density-dependent effects on growth, either via population-level effects or via intraspecific competition among individuals. There was little evidence of density-dependent effects on fecundity but few studies examined this relationship. Most studies that examined population-level effects demonstrated density-dependence; however, these studies were not common, likely because they require long time series of data (at least 3-4 times mean generation time for the species) to be rigorous tests.

Stock-recruitment analyses for unexploited populations were uncommon and showed both evidence and a lack of evidence for density-dependent effects. The most common density-independent effect was manifested via the effects of unpredictable environment effects that sometimes lasted multiple years (e.g., floods, droughts, extreme temperature events) and were particularly evident on fish in their first year of life. There was no evidence that the frequencies of density-dependent or density-independent effects differed among the nine species examined, but the number of studies per species varied from 2 to more than 20, hence, this conclusion is tentative. In addition, there were few studies that examined the effects of density on fecundity and survivorship; therefore, our evaluation of the lack of density-dependent responses on these factors is more likely limited by a lack of data rather than an absence of this phenomenon in salmonid populations. The predictions of general climate models all yield an increasing probability that unpredictable environmental events will increase in frequency and this will likely affect salmonid populations via increased impacts of density-independent effects. It is possible that these impacts could become so strong or frequent that some salmonid populations will reach a point where compensatory density-dependent responses will no longer be possible. This may result in loss of populations in marginal habitats or populations that persist at low density such as some species or populations of concern. One positive effect of the ubiquity of density-dependence in growth, could be the creation of trophy salmonid fisheries via selective manipulation of population density, especially of smaller fish.



Where did all the Trout go? Combining Multiple Measures of Fish Movement to Gain Insights into Brook Trout Population Connectivity

Shannon L. White^{1,2}, Stephanie A. Dowell³, Meredith L. Bartron³, and Tyler Wagner⁴

¹Pennsylvania Cooperative Fish and Wildlife Research Unit, Pennsylvania State University, University Park, USA

²Department of Ecosystem Science and Management, Pennsylvania State University, University Park, USA

³U.S. Fish and Wildlife Service, Northeast Fishery Center, 227 Washington Avenue, Lamar, Pennsylvania 16848, USA

⁴U.S. Geological Survey, Pennsylvania Cooperative Fish and Wildlife Research Unit,
Pennsylvania State University, University Park, USA

Extended Abstract

Understanding movement dynamics - the distribution of individuals over space and time - is critical to effectively manage and conserve populations of stream salmonids. However, despite decades of research, there remains considerable uncertainty about the processes that effect movement of salmonids. Movement in many populations is leptokurtotically distributed, with most individuals remaining relatively sedentary and only a small proportion of the population moving long distances. However, significant variance in movement behavior within and among populations suggests the potential for cross-scale interactions that influence individual fish movement at different temporal and spatial scales. For example, intraspecific movement rates may vary along the riverscape and fluctuate differently in response to time of year and interannual variation in climactic variables. Failure to accurately describe species' movement patterns can result in improper management decisions that decrease connectivity and make populations more susceptible to local extirpation. Understanding movement dynamics may be particularly important in systems where coldwater tributary connectivity is maintained through seasonally suitable main-stem habitat.

To elucidate temporal and spatial patterns in Brook Trout *Salvelinus fontinalis* movement, we studied four populations of native Brook Trout in the Loyalsock Creek watershed in Pennsylvania using radio telemetry and variable microsatellite loci to quantify genetic variation. The combination of methods allows for the study of individual-specific movement at high spatial and temporal resolution, while also determining the long-term effects of movement on population connectivity and gene flow. The main stem of Loyalsock Creek is a shallow, fourth-order tributary with stream temperatures that exceed Brook Trout thermal tolerance in summer.

As such, Loyalsock Creek is not viable Brook Trout habitat from approximately June to September. In 2016, we tagged 180 adult Brook Trout in spring and late summer in four first- and second-order tributaries that flow directly into Loyalsock Creek. We tracked fish movements twice a week from May to November to document individual movements and quantify behavior as a function of stream flow and time of year. To determine the influence of movement on population connectivity, we genotyped at least 50 fish from each of seven adjacent streams that flow directly into Loyalsock Creek, including the four streams where we completed the telemetry study.

Telemetry results revealed considerable differences in individual movement within and across populations. While the majority of fish across all four tributaries remained sedentary during summer months, a significant proportion of fish at two sites dispersed several kilometers downstream into Loyalsock Creek after spawning in fall. Fish in the other tributaries showed no temporal variation in behavior, with overall low rates of movement throughout the tracking period. Results from population genetic analyses suggest at least moderate levels of connectivity in the populations studied. Genetic assignment tests revealed several fish that were captured outside of their natal stream, including fish that we tracked into Loyalsock Creek. We did not document tributary-to-main stem movement at two sites during telemetry tracking; however, these sites had similar allelic frequencies with adjacent tributaries suggesting that, at least historically, there was immigration and/or emigration at these sites. Because movement into the main stem was not documented with telemetry, connectivity at these two sites may have recently been lost, or patterns in fish movement could vary over broader time scales than could be detected during the scope of this telemetry study.

Brook Trout populations have the potential to be highly isolated, with limited gene flow occurring at even small spatial scales. However, our results highlight the need to consider multiple temporal and spatial scales when designing and interpreting trout movement studies. In particular, genetic results indicate varying levels of connectivity among populations separated by several kilometers and a large main-stem river channel with atypical Brook Trout habitat. Long-distance dispersal events into Loyalsock Creek likely maintain population connectivity through at least intermittent gene flow, and suggests that the main stem is critical Brook Trout habitat that is used as a seasonal movement corridor. Significant

variance in individual movement behavior suggests the potential for multiple life history strategies within the Brook Trout populations monitored. As mobile individuals are less susceptible to the lethal effects of local habitat loss and help to maintain population connectivity (including the ability to potentially recolonize habitat), our results highlight the need to consider phenotype-specific behavior in conservation and management plans. Failure to do so could limit the ability to conserve phenotypes that contribute towards the maintenance of connectivity and genetic diversity in Brook Trout metapopulations. Further work is needed to determine the cause of individual variation in behavior.



Long-term Effectiveness of Flow Management and Fish Passage on the Henrys Fork Rainbow Trout Population

Bryce Oldemeyer¹, Jon Flinders², Christina Morrisett³, Rob Van Kirk⁴

¹Research Associate, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, FAX 208-652-3568, bryce@henrysfork.org.

²Regional Fisheries Biologist, Idaho Department of Fish and Game, 4279 Commerce Circle, Idaho Falls, ID 83401, 208-525-7290, jon.flinders@idfg.idaho.gov.

³Research Assistant, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, christina.morrisett@gmail.com.
Present address: School of Aquatic and Fisheries Sciences, University of Washington, Seattle, WA, 98195.

⁴Senior Scientist, Henry's Fork Foundation, 512 Main St, Ashton, ID 83420, 208-652-3567, rob@henrysfork.org.

Abstract—The reach of the Henrys Fork of the Snake River immediately downstream of Island Park Reservoir is a world-renowned fishery. Its productive water supports high densities of macroinvertebrates and an abundant population of quality-sized, and larger, Rainbow Trout *Oncorhynchus mykiss*. Rainbow Trout are not native to the Henrys Fork and were stocked from the early 1900s through 1977. In 1978, stocking ceased, and the Rainbow Trout population has been managed under wild trout regulations. Rainbow Trout abundance declined from 3,643 fish/km to 2,528 fish/km between 1978 and 1994 and has averaged 1,857 fish/km since then. In this study, we summarize results of ongoing research that began in 1994 to identify factors limiting recruitment to the wild Rainbow Trout population. Habitat availability for juvenile trout, particularly during winter months, was the primary factor limiting recruitment. Projects reconnecting tributary habitat and increasing river flow during winter months increased juvenile trout habitat and increased recruitment an estimated 9%. Unfortunately, lack of available water during dry years regularly limits winter flows under constraints of filling irrigation rights in the reservoir. Given hydrologic trends caused by a warming climate, the wild Henrys Fork Rainbow Trout population may not be able to attain abundances comparable to those during decades when the fishery was stocked.

Introduction

The reach of the Henrys Fork of the Snake River immediately downstream of Island Park Reservoir is a world-renowned fishery. Its productive water supports high densities of macroinvertebrates, and subsequently, an abundant population of Rainbow Trout *Oncorhynchus mykiss* (Van Kirk and Gamblin, 2000). Starting in the early 1900s to 1977, nonnative Rainbow Trout were stocked in this reach of the Henrys Fork River, and Rainbow Trout densities averaged 3,643 fish/km (Figure 1). In 1978, stocking ceased and the Rainbow Trout fishery was managed as a wild population. Rainbow Trout stocked in Island Park Reservoir regularly supplemented the river population below the dam and Rainbow Trout densities average 2,528 fish/km from 1978 to 1994. In 1994, a hydroelectric facility with fish screens was installed on the dam, greatly reducing opportunities for fish passage to the river downstream, and average Rainbow Trout densities dropped to 1,857 fish/km.

Since 1994, intensive research has been done to identify bottlenecks within the population. In the late 1990s, habitat availability for juvenile Rainbow Trout, particularly during winter months, was identified as the factor most likely limiting juvenile survival and recruitment (Meyer 1995; Gregory 2000; Mitro et al. 2003). To increase available habitat and juvenile recruitment, a fish ladder was renovated in 2005 to bolster juvenile fish passage to a major tributary of the Henrys Fork downstream of Island Park Dam (Buffalo River). Beginning in the winter of 2005-2006, a multi-stakeholder committee called the Henry's Fork Drought Management Planning Committee (DMPC) began managing winter fill of Island Park Reservoir to maximize outflow during the coldest part of the winter, December 1 through February 28, under the constraint of filling storage rights in the reservoir (Joint Committee 2005).

In this study, we quantified the benefits of the renovated Buffalo River fish ladder and increased winter flows on the Henrys Fork Rainbow Trout

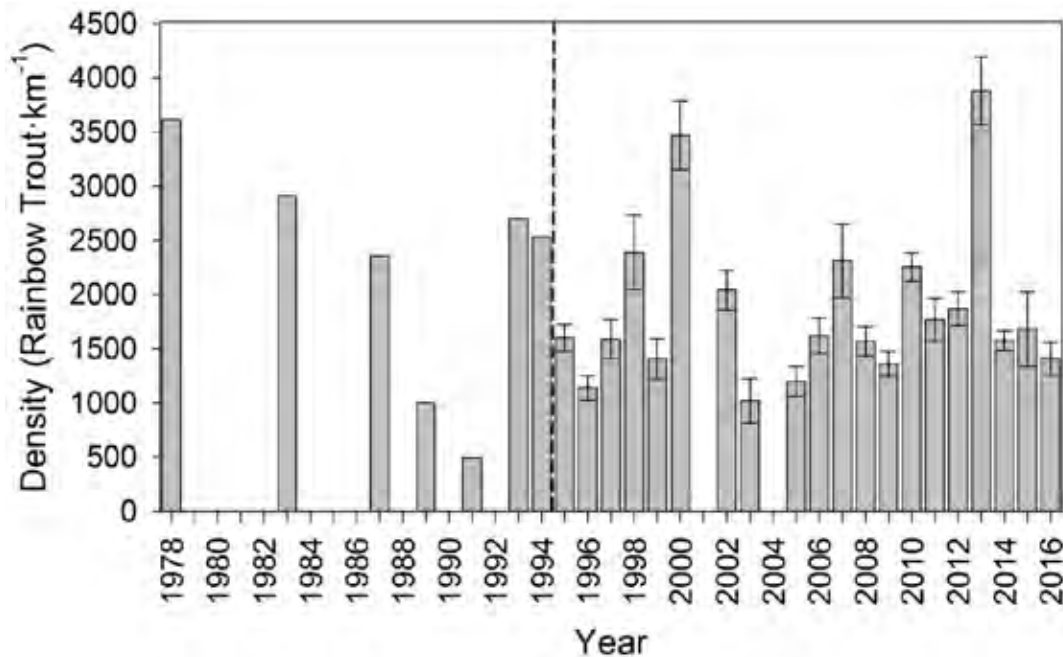


Figure 1. Rainbow Trout per kilometer with 95% confidence interval for the reach of the Henrys Fork River directly below Island Park Dam from the time wild-trout regulations were first implemented to 2016. Dashed line between years 1994 and 1995 signifies when a fish screen was installed on the hydroelectric facility at Island Park Dam and intensive monitoring of the Rainbow Trout population began.

population below Island Park Dam. We used 10 years of data from 2006 to 2016 collected at the Buffalo River fish ladder to quantify juvenile Rainbow Trout winter use of the Buffalo River and contribution back to the Henrys Fork River Rainbow Trout population. Flow data from Island Park Dam, Buffalo River, and population estimates obtained from multi-pass electrofishing in the reach below Island Park Dam from 2006 to 2016 were used to estimate trout abundances for a theoretical scenario where flows were not increased from December 1 through February 28 and compared those abundance estimates to those using actual flows during the same time period.

Study Site

The Henrys Fork of the Snake River is roughly 180 km long and located in southeastern Idaho near the Idaho, Montana, and Wyoming border (Figure 2). The most renowned section of the Henrys Fork is the 34.4-km reach immediately downstream of Island Park Dam. This reach supports high densities of macroinvertebrates (~45,000 individuals/m²), a mixture of pocket water in two canyon sections, and a

meandering flat-water section through Harriman State Park. Abundant insects and diverse habitat provides both wade fisherman and boat fisherman unique opportunities to sight-fish large, Rainbow Trout with dry flies during most of the fishing season (Lawson 2012; McDaniel 2012).

Between 1936 and 1938, Island Park dam was constructed for irrigation storage roughly 30.5 km downstream from the Henrys Fork River headwaters (Figure 2). To build Island Park Dam, a hydroelectric facility was built on the Buffalo River, a major tributary to the Henrys Fork, approximately 0.5 km downstream from Island Park Dam. These two projects severely limited access for Rainbow Trout to headwater spawning and overwinter habitat. The effects of decreased natural reproduction were mitigated by stocking trout until 1977. In 1994, a fish screen was installed on the newly constructed Island Park Dam power plant intake, limiting opportunities for downstream migration of wild and hatchery fish from the reservoir into the river. Fremont-Madison Irrigation District holds irrigation rights to all of the water stored in Island Park Reservoir, and the reservoir must be filled before irrigation season each year to

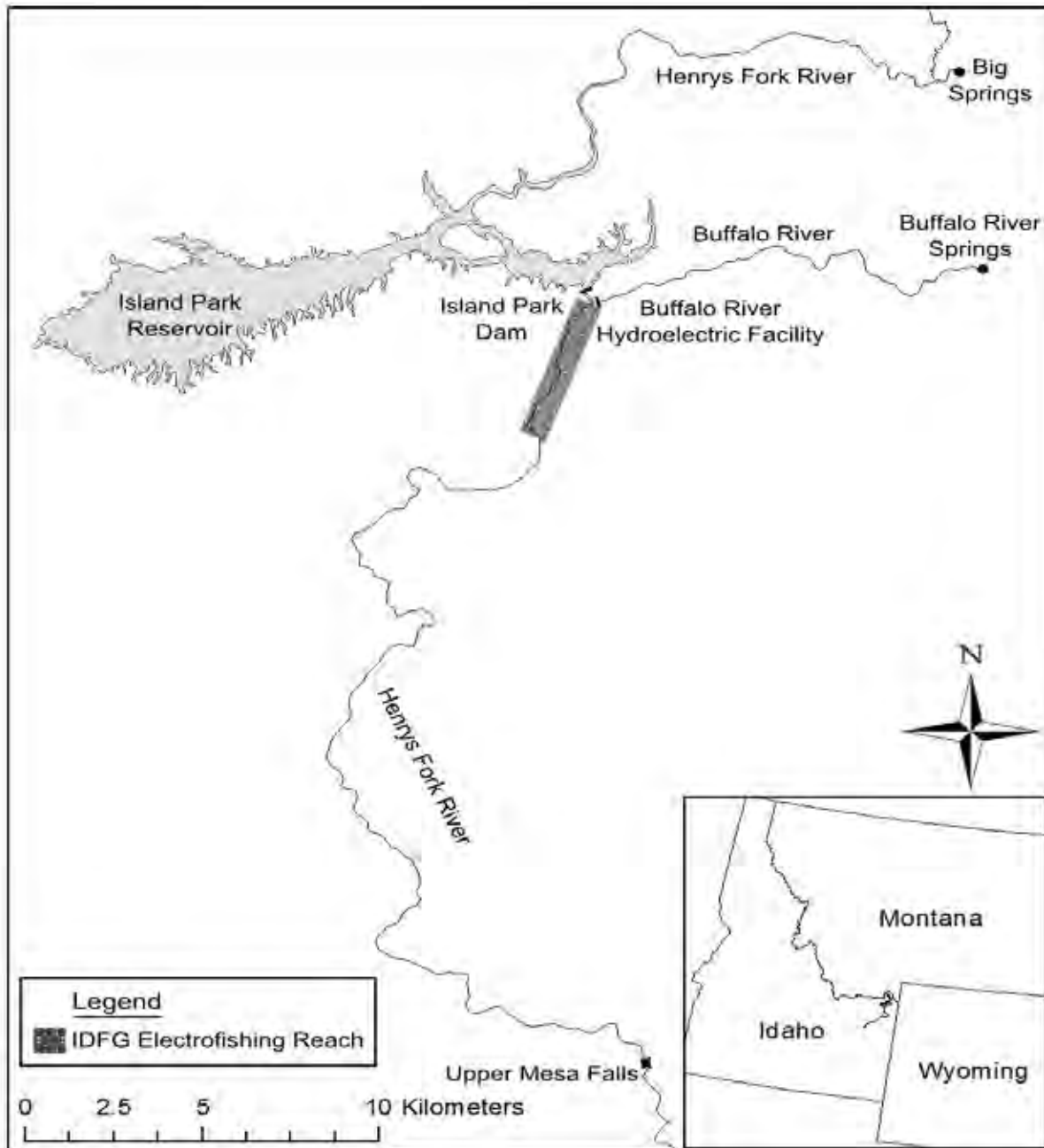


Figure 2. Map of the Henrys Fork of the Snake River downstream of Island Park Dam, Idaho.

meet these rights. Water is generally stored between October and April, and the amount of water that can be released during the storage season is primarily a function of carryover from the previous irrigation season (Benjamin and Van Kirk 1999).

Unlike Island Park Dam, the Buffalo River hydroelectric facility was built with a fish ladder that was designed to promote adult trout access to spawning habitat. Unfortunately, the gradient of

the fish ladder was too high to allow juvenile trout access to prime overwinter habitat on the Buffalo River. Due to declines in Rainbow Trout abundances in the Henrys Fork below Island Park Dam post-1994, relicensing for the Buffalo River facility in 2004 was conditional on renovating the fish ladder to accommodate juvenile trout passage. In 2006, the Buffalo River fish ladder renovation was completed and intensive Rainbow Trout monitoring began.

Methods

Juvenile Rainbow Trout Abundance Estimate

From 2006 to 2016, the Idaho Department of Fish and Game (IDFG) conducted annual multi-pass electrofishing surveys in a 4-km reach that begins 0.45 km downstream of Island Park Dam (Figure 2). This stretch was identified as the best overwinter rearing and spawning habitat below Island Park Dam and was used as an index reach to monitor trends in the Rainbow Trout population.

Three rafts outfitted with electrofishing equipment were used to conduct three to four sampling events each May. All trout collected during mark-recapture surveys were identified to species and measured for total length (TL, mm). Those exceeding 150 mm were marked with a hole punch in the caudal fin prior to release. In all reaches, we estimated abundance for all trout > 150 mm using the Log-likelihood method in Fisheries Analysis+ software (Montana Fish, Wildlife, and Parks).

Flow Recruitment Model

In 2006, Garren et al. (2006) found that mean December-February flow from Island Park Dam during a cohort's first winter was the best predictor of age-2 Rainbow Trout abundances in the reach below Island Park Dam. Prior to formal establishment of DMPC, October-March outflow from Island Park Dam was generally set at the constant flow rate that would achieve a given April-1 reservoir volume (long-term average of 83% full on April 1). We considered this the "baseline" fill strategy. Starting with the winter of 2005-2006, the DMPC implemented a new fill strategy in which outflow was lowered in October and November to store water at a higher rate during this time period so that the same April-1 target could be achieved with higher outflow from December through February.

In 2016, we found that adding mean flow from the Buffalo River to the outflow of Island Park Dam increased the predictability of the earlier model (Figure 3). This updated age-2 Rainbow Trout abundance model was used to calculate expected age-2 abundance for recruitment years 2007-2016 under the actual fill strategy implemented by the DMPC over winters 2005-2006 through 2014-2015.

The same model was then used to calculate expected age-2 abundance for the same set of years under a hypothetical flow scenario that assumed application of the previous baseline fill strategy. The two recruitment estimates were then compared to obtain the expected difference in age-2 abundance between the baseline fill strategy and the new DMPC strategy.

Buffalo River Fish Ladder

Continuous trapping of fish utilizing the renovated Buffalo River fish ladder began March 2006. The trap was checked three times per week and fish were measured and identified to species before being released upstream of the Buffalo River hydroelectric facility. Due to declines in fish passage, the trap screen was removed and the trap was not operated from July 1 to August 31 and December 1 to February 28 starting in 2013.

In 2013 and 2014, a sample of juvenile Rainbow Trout migrating up the Buffalo River fish ladder in September 1 through December 31 were implanted with 12-mm long Passive Integrated Transponder (PIT) tags. These passively operating tags provide a unique 13-digit code that could be read when the tagged fish passed an electromagnetic antenna or handheld reader. A pit tag antenna was installed on the major spillway of the Buffalo River and temporal data were recorded for PIT tagged fish emigrating out of the Buffalo River that had been marked and passed upstream in 2013 and 2014.

To calculate the contribution that the renovation of the Buffalo River fish ladder had to juvenile Rainbow Trout passage and benefit to the Henrys Fork fishery, we first calculated apparent survival for individuals in 2013 and 2014. We did this by dividing the number of fish detected at the Buffalo River fish ladder PIT tag antenna in 2014 and 2015 by the number of fish passed upstream in 2013 and 2014. We then divided that rate by 50% to incorporate presumed mortality that would have taken place before the individual reached age 2 in the Henrys Fork River.

We applied the apparent survival rate to the number of juvenile Rainbow Trout passed upstream each fall from 2006 through 2014. For years 2006 and 2007, we used the mean number of juvenile Rainbow Trout passed upstream during all the other years to supplement the missing data.

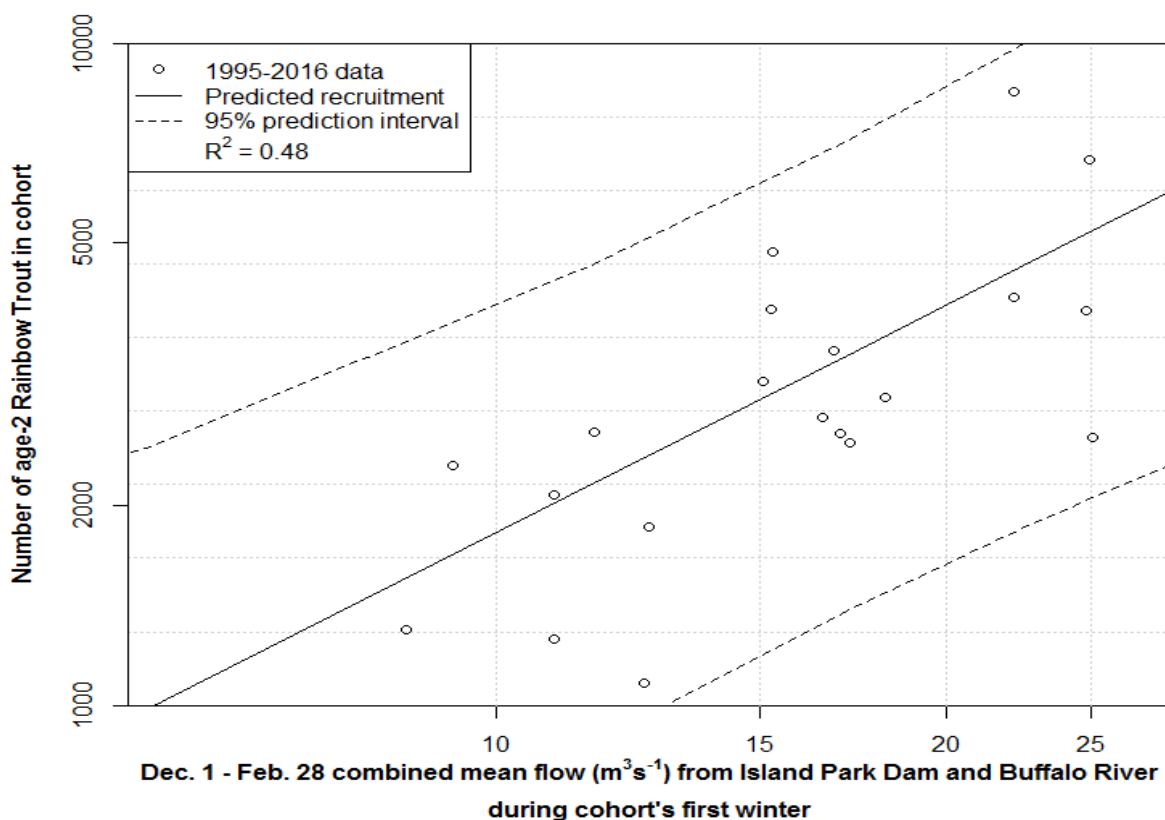


Figure 3. December 1 through February 28 mean flow (m^3s^{-1}) from Island Park Dam and Buffalo River during a cohorts first winter and corresponding abundance of that cohort at two years of age, 1995-2016.

Results

From 2006 to 2016, between 639 and 1532 Rainbow Trout were marked, and electrofishing efficiencies were between 0.09 and 0.28. Mean abundance of age-2 Rainbow Trout was 3,329. Maximum age-2 abundance was 8,747 in 2012 and minimum age-2 abundance was 1,078 in 2015.

Implementing the management practice of increasing flows from December 1 through February 28 increased flows an average of $0.679 \text{ m}^3\text{s}^{-1}$ (5% flow increase) from 2006 to 2016 compared to the baseline scenario in which flows remained constant throughout the storage season. Using the age-2 abundance estimate model with combined mean flows from Buffalo River and Island Park Dam from December 1 through February 28 as the sole predictor variable, increasing flows during December 1 through February 28 added an average of 140 Rainbow Trout, or 5.7%, annually to the total population (Figure 4).

From 2006 to 2016, an average of 1,898 juvenile Rainbow Trout migrated up the Buffalo River fish ladder annually from September 1 through December 1. In 2013 and 2014, 1,605 and 1,788 juvenile Rainbow Trout were implanted with PIT tags as they were passed upstream of the Buffalo River hydroelectric facility. In 2014 and 2015, 105 and 152 PIT tagged fish were detected emigrating from the Buffalo River to the Henrys Fork in the spring. After applying a 50% mortality rate to account for the time between emigrating from the Buffalo River to becoming two years of age in the Henrys Fork River, apparent survival for juvenile Rainbow Trout migrating up the Buffalo River in the fall and returning to the Henrys Fork to reach age 2 was 3.8%. Expanding this rate to the mean number of juvenile Rainbow Trout that migrate up the Buffalo River fish ladder in the fall, the Buffalo River fish ladder contributed 72 fish, or 3%, annually to the total Henrys

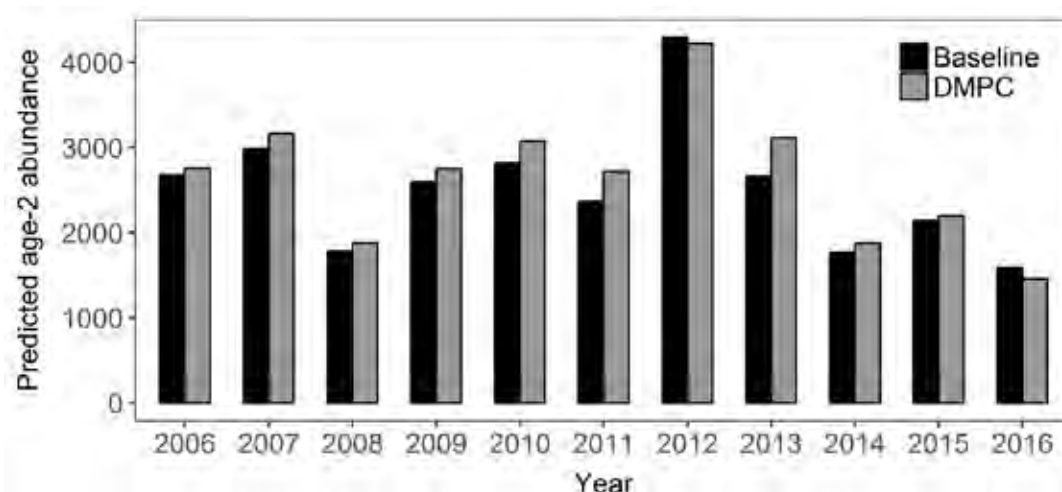


Figure 4. Predicted age-2 Rainbow Trout abundances below Island Park Dam for a hypothetical baseline fill strategy (Baseline) where flows from Island Park Dam, Idaho, were not increased during a portion of the winter compared to predicted age-2 Rainbow Trout abundances using observed flows implemented by the Drought Management Planning Committee (DMPC), 2006-2016.

Fork River Rainbow Trout population below Island Park Dam.

Discussion

Increased flows from Island Park Dam from December 1 through February 28 provided slight increases to the Henrys Fork fishery. The amount of water available to increase flows during the winter was largely restricted by the previous year's snowpack, runoff timing, and irrigation demand. If natural river flows were able to satisfy irrigation rights and demand, surplus water stored at Island Park during the summer and fall could be released during the winter to increase juvenile Rainbow Trout survival. Unfortunately, springtime temperatures in the Henrys Fork watershed have increased an average of 2.48 °C from 1989-2016 (Van Kirk 2017). This has led to earlier peak runoff, requiring increased use of storage water out of Island Park Dam during the summer, and decreasing the amount of storage water available for winter flows. In addition, the variability of snowpack and runoff timing has increased since the 1980s and it is unlikely that there will be a series of consecutive years of increased winter flows needed to achieve Rainbow Trout abundances comparable to those during the mid-1900s when the fishery was stocked.

Renovating the Buffalo River fish ladder to increase juvenile Rainbow Trout access to overwinter

habitat provided only modest increases to the Henrys Fork fishery. The renovation allowed juvenile Rainbow Trout access to overwinter habitat and a 3% increase of Rainbow Trout in the Henrys Fork population. The 3% increase to the population is likely a conservative estimate due to the 50% mortality rate used to account for the time between emigrating out of the Buffalo River and turning two years of age in the Henrys Fork River. Trout survival is highly variable between systems and seasons but typically increases substantially after age-1 (Rieman and Apperson 1989; Carlson and Letcher 2003). We chose to use the 50% survival rate as to not overestimate the Buffalo River juvenile Rainbow Trout contribution.

The renovated Buffalo River fish ladder had additional benefits to the Henrys Fork Rainbow Trout population outside of increased access to overwinter habitat. The lower gradient of the renovated fish ladder increased passage of adult Rainbow Trout migrating to spawn in Buffalo River (Morrisett 2016). In addition, preliminary genetic analysis shows that some offspring of Henrys Fork Rainbow Trout that spawn with Buffalo River Rainbow Trout emigrate and contribute to the Henrys Fork population. Upon completion of the genetic analysis, we hope to be able to quantify the total benefit that the Buffalo River fish ladder, and Buffalo River Rainbow Trout population, have to the Henrys Fork Rainbow Trout population.

In summary, these two projects were successful at increasing the Henrys Fork Rainbow Trout population below Island Park Dam. Unfortunately, the combined total effect of the two management actions was only a 9% increase in the Rainbow Trout population with an interannual coefficient of variation of 38% over the 1995-2016 time period. With increasing regional temperatures, and without substantial decreases in irrigation demand, it is unlikely that the iconic Henrys Fork fishery below Island Park Dam will be able to attain Rainbow Trout abundances comparable to those during decades when the fishery was stocked.

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Movement of Coldwater Fishes in an Un-fragmented Watershed in Central Montana

Michael Lance¹, Al Zale², Tom McMahon³, Jason Mullen⁴, Grant Grisak⁴, and Robert Al-Chokhachy⁵

Affiliations: ¹Montana Cooperative Fishery Research Unit, PO Box 173460, Montana State University, Bozeman, MT 59717-3460

²US Geological Survey, Montana Cooperative Fishery Research Unit, PO Box 173460, Montana State University, Bozeman, MT 59717-3460

³Montana State University, Ecology Department, PO Box 173460, Montana State University, Bozeman, MT 59717-3460

⁴Montana Fish, Wildlife and Parks, Region 4 Fisheries, 4600 Giant Springs Road, Great Falls, MT 59405

⁵US Geological Survey, Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, MT 59715
-Author contact: Michael Lance, michael.j.lance@gmail.com

Introduction

Movement is a key component of the life history of fishes as it provides individuals with access to diverse resources and habitats (Lucas and Baras 2001). Fish move for a wide variety of reasons, the most important of which are spawning, feeding, and refuge (Schlosser 1991). The spatial and temporal availability of spawning, feeding, and refuge habitats can structure the movement and presence of fish populations in diverse riverscapes (Torgersen et al. 2006). Although movement is associated with more risk, mobile individuals generally have higher individual fitness and reproductive potential compared to resident individuals (Brönmark et al. 2014). The extent to which fish move sets the spatial boundaries of populations. Moreover, movements of individuals can strengthen the viability of metapopulations by facilitating recovery from localized disturbances. Fisheries and land managers need to be aware of the movement patterns of fish to apply management and conservation actions at appropriate scales.

Study Area and Methods

The Smith River in central Montana flows 193 km and then joins the Missouri River near Great Falls, Montana. The Smith River is unique in that there are no permanent barriers to fish movement in the main stem or its major tributaries. The Smith River flows through three major geomorphic regions: a wide headwater valley that is bounded by mountains, a deeply incised limestone canyon in the middle portion of the watershed, and a low gradient prairie reach. The Smith River supports a self-sustaining coldwater fish assemblage, mostly made up of Brown Trout *Salmo trutta*, Mountain Whitefish *Prosopium williamsoni*, and Rainbow Trout *Oncorhynchus mykiss*. Anglers

come from across the United States and the globe to float through the picturesque canyons of the Smith River and pursue its abundant trout and whitefish. Seasonally, some fish in the Smith River migrate in excess of 100 km (Grisak et al. 2012). To improve our understanding of the movement behaviors of fish in the Smith River watershed, we marked over 7,500 fish with half-duplex passive integrated transponder (PIT) tags and monitored their movements past 14 stationary PIT tag readers. We also located fish with mobile PIT tag readers. Because fish had to move to be detected at stationary readers, detection probability at stationary readers was used as an indicator of the propensity for groups of fish to move. Detection probability was calculated as the proportion of tagged fish that were detected at least once at a stationary PIT tag reader. Detection probability was analyzed using generalized linear models and a binomial distribution where predictor variables were species, tagging location, and total length. The response variable was probability of detection at a stationary reader. We analyzed the probability of detecting fish moving among both tributary and main-stem habitats as well as the probability of detecting fish moving among more than one of the Smith River's three major geomorphic regions. Although the Smith River hosts 12 taxa of fish, we restricted detection probability analyses to the three most abundant species: Brown Trout, Mountain Whitefish, and Rainbow Trout.

Individual travel routes were analyzed using ArcGIS 10.4 Network Analyst tools (ESRI 2015, ArcGIS Desktop: Release 10.4.1, Redlands, CA: Environmental Systems Research Institute). Routes created with the Network Analyst tools depicted the stream network travelled by tagged fish as they moved throughout the watershed. The number of

tagged fish traversing a stream segment was summed to identify where fish moved most often in the stream network. Each pass of a tagged fish through a stream segment was weighted by the number of fish in the tag group of the passing fish. A tag group represented the total number of a species of fish tagged in a reach. Reaches were delineated by major tributary junctions, readers, and geomorphic region transitions. For example, 43 Brown Trout were tagged in the Smith River from Tenderfoot Creek downstream to the Castle Bar stationary reader and constituted one tag group. Similarly, 197 Mountain Whitefish, and 54 Rainbow Trout were tagged in the same reach and each constituted a tag group. If thirty of the tagged Brown Trout moved through a reach of stream, their movements would represent 30/43 of a tag group or 0.69 tag group equivalents (TGE). If 15 Mountain Whitefish moved through the same stream reach, the total movement through that stream reach in TGE would be 30/43 (0.69 TGE) + 15/197 (0.08 TGE) and would result in a total of 0.77 TGE. If fish from multiple tag locations passed through a stream reach, they were also summed into the TGE value for that reach. If 180 of the 362 Rainbow Trout tagged in the Smith River near Sheep Creek passed the same reach in the previous example, the TGE value of movement past that reach would increase by 180/362 (0.50 TGE) resulting in a total for the example reach of 1.27 TGE. This process was repeated for all fish from all species and tag locations. However, tag groups with less than 10 individuals were excluded from our analyses because movements of fish from these groups would have an unrepresentative influence on TGE values. The sum of TGEs provided a quantitative measure of the relative number of tagged fish from multiple species and locations that moved through a stream reach. This method identifies stream reaches that are important corridors for maintaining the diversity of species and life history patterns of fish in the Smith River.

Results

Movement was documented throughout the Smith River watershed, and some fish left the Smith River watershed and entered the adjacent Missouri River watershed. Movements of tagged fish were observed across 474 km of the Smith River and Missouri River watersheds. Within the Smith River watershed, the magnitude of movement varied and was greatest along the main stem (Figure 1). The highest amount of

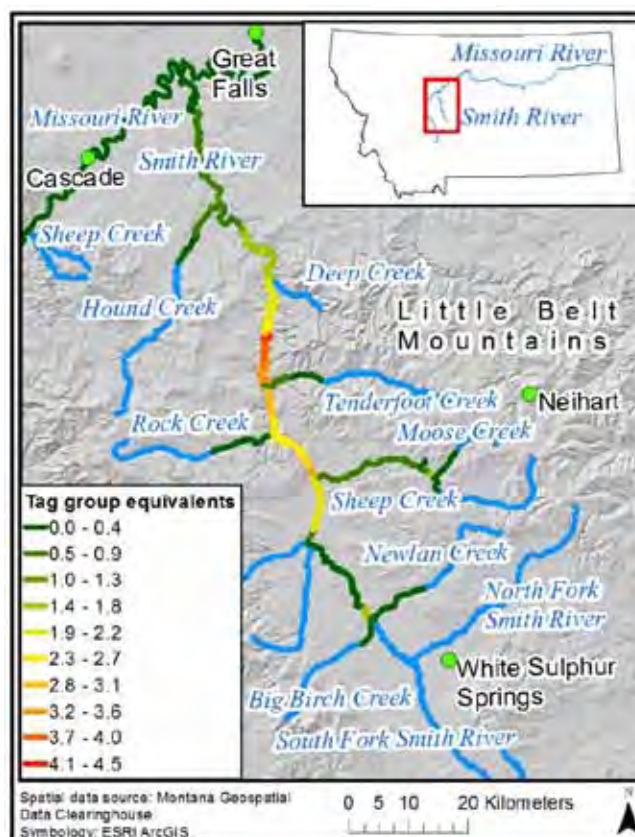


Figure 1. Movement patterns of fish tagged in the Smith River watershed. Orange and red lines represent portions of the stream network where movement was observed most often. Green lines represent stream reaches where movement was observed least often. Yellow lines are stream reaches with more movement compared to green lines but less than areas marked in orange or red. Blue lines represent streams outside of our study area.

movement was observed immediately downstream of the confluence with Tenderfoot Creek and along the Smith River about 18 km downstream. Movement was also common in the Smith River downstream of the confluence with Sheep Creek. Movement was most common in the canyon geomorphic region compared to the prairie and headwaters geomorphic regions. The magnitude of fish movement in the lower portions of some tributaries and along the length of Sheep Creek was equivalent to portions of the main-stem Smith River. However, in tributaries, movement in upstream reaches was less common than in downstream reaches.

Mountain Whitefish moved most often, followed by Rainbow Trout; Brown Trout showed the highest degree of residency. Fish that moved were on average

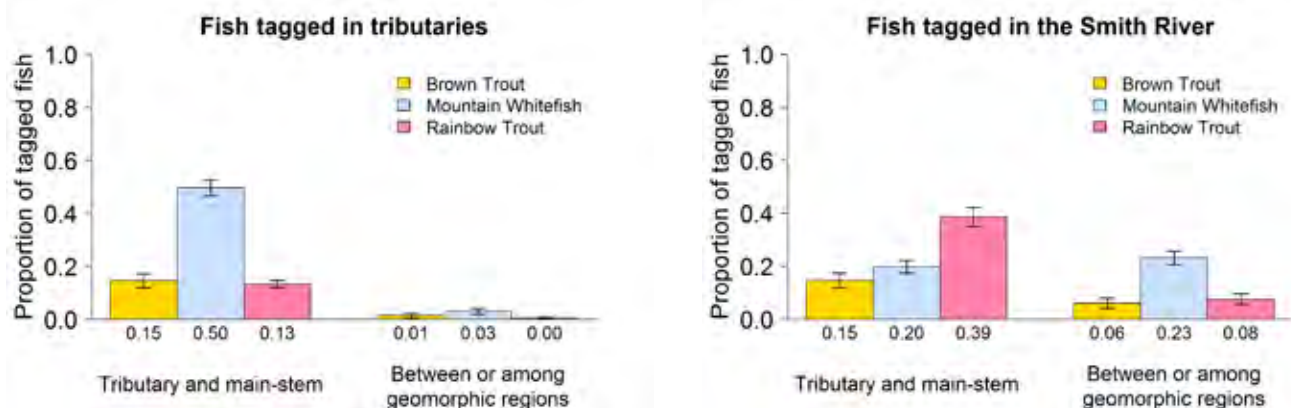


Figure 2. Proportions of tagged fish that used multiple habitat types. The left panel depicts fish tagged in tributaries, and the right panel depicts fish tagged in the main-stem Smith River. Error bars represent 95% confidence intervals. Numbers below bars are the numeric proportion of fish detected in each category.

32 mm longer than fish that did not move. However, the mass at length of Mountain Whitefish that moved was 3% less than that of individuals that did not move, indicating that when compared to resident individuals, mobile Mountain Whitefish had a more slender body shape or poorer body condition. The proportion of fish that used both tributary and main-stem habitats as well as the proportion of fish that used multiple geomorphic regions varied by species and where fish were tagged (Figure 2). Mountain Whitefish tagged in tributaries were more likely to use main-stem habitats than other species tagged in tributaries. Of fish tagged in the main-stem Smith River, Rainbow Trout were most likely to enter tributaries. Mountain Whitefish were most likely to use multiple geomorphic regions compared to Rainbow Trout and Brown Trout. Fish tagged in tributaries were less likely to use multiple geomorphic regions compared to fish tagged in the main-stem.

Fish movement varied throughout the year. Peak movements were observed during species-specific spawning seasons, and substantial non-spawning movements were observed during the spring and autumn. Non-spawning movements in spring and autumn were numerically about equivalent to spawning movements and were probably associated with feeding and outmigration. We also observed movements of all species occurring near the summer solstice concurrent with the onset of summer base flows. The seasonal timing of observed movements

supports the “spawning, feeding, refuge” paradigm of fish movement (Schlosser 1991).

Discussion and Conclusions

Consideration of the variability in movement patterns among and within species is beneficial to the conservation and protection of coldwater fish assemblages. Conservation measures that maintain connectivity among stream networks and that incorporate life history information on all components of a fish assemblage will preserve the diversity of expressed life history patterns, thereby enhancing fishery productivity, sustainability, and resilience. Preserving and restoring watershed connectivity is particularly important near confluences of major tributaries because these stream reaches can be movement corridors for spawning, feeding, and refuge. Timing of movements also needs to be considered in the management of stream connectivity. Connectivity is critically important during species-specific spawning periods as well as during the spring and autumn. Additionally, connectivity is important during early to mid-summer when fish are probably moving to habitats with cooler water. The diversity of movement patterns expressed by the Smith River fish assemblage undoubtedly has contributed to the persistence and resilience of this important fishery. Our results clearly show that the protection of movement corridors is essential for conserving the coldwater fish assemblage in the Smith River and other similar inland watersheds.

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Overwintering Habitat use by Westslope Cutthroat Trout in Mountain Headwater Streams of Southern Alberta, Canada

Jeremy W. Benson¹, Andreas Luek², Joseph B. Rasmussen³

University of Lethbridge, 4401 University Dr W, Lethbridge, AB, Canada, T1K 3M4,
Water and Environmental Science Building.

Email: 1jw.benson@uleth.ca; 2andreas.luek@uleth.ca; 3joseph.rasmussen@uleth.ca

Abstract—Westslope Cutthroat Trout (WCT) *Oncorhynchus clarkii lewisi* are a threatened species in Alberta, Canada. Most remaining WCT populations are restricted to small headwater streams along the Rocky Mountains' eastern slopes. Winter is often a critical time, where anthropogenic effects such as land-use and resource extraction may have amplified effects on the survival of WCT populations. Inaccessibility of the headwater stream habitat during winter and limitations of cold tolerant equipment has hindered detailed winter studies so far. Non-invasive survey techniques, such as snorkeling, underwater cameras, and comprehensive habitat surveys were used to identify key features of WCT overwintering habitat quality, quantity and use by WCT. Pools were identified as primary overwintering habitat for adult WCT. While WCT use of pools is variable, WCT were observed in greater abundance in suspected high quality overwintering pools. An overall decrease in abundance was observed from summer to winter, mainly due to the disappearance of smaller size classes. Larger size class abundance was strongly related to water temperature and ice cover. We concluded that overwintering pools are of increased importance for winter survival of WCT in small, high gradient mountainous streams, because habitat fragmentation and generally small habitat size will cause decreased movement between habitats.

Introduction

Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) are listed as threatened under the Canadian Federal Species at Risk Act in Alberta, Canada (The Alberta Westslope Cutthroat Trout Recovery Team 2013). The reduction in WCT populations across Alberta is mainly due to habitat loss and alteration, as well as introgression by Rainbow Trout *Oncorhynchus mykiss* (The Alberta Westslope Cutthroat Trout Recovery Team 2013). Currently, genetically pure WCT populations only occupy approximately 5 % of the historic range in Alberta (Mayhood and Taylor 2011). Many of the remaining Alberta WCT populations have been restricted to small, high elevation headwater streams along the Rocky Mountains' eastern slopes.

Overwintering habitat has been identified as a crucial feature of adult WCT survival, especially for the remaining pure WCT populations in southwestern Alberta (Fisheries and Oceans Canada 2014). There is limited knowledge on overwintering habitat for stream dwelling salmonids in general, but specifically for WCT. The Alberta Westslope Cutthroat Trout Recovery Team (2013) has identified this knowledge gap and included a call for research on WCT

overwintering habitat in their habitat protection plan. Specifically, research is needed to identify overwintering habitat attributes and how the habitat is utilized by WCT (Fisheries and Oceans Canada 2014). It is further unclear how anthropogenic effects such as land-use and resource extraction influence the survival of WCT populations during winter and within a larger frame of climate change.

Recently, studies have identified groundwater and different types of ice as major determinants of habitat suitability during winter (Brown and Mackay 1995; Brown 1999). Ice conditions such as frazil and anchor ice reduce habitat availability (Brown and Mackay 1995) and can be directly harmful to fish (Brown et al. 1994). Frazil ice being ice crystals formed in supercooled water too turbulent to freeze solid, which is then transported through the water column (Huusko et al. 2007). Anchor ice is also formed due to supercooled water, appearing on the stream bed usually in shallow turbulent habitats. Large numbers of fish transition from territorial to gregarious behaviour in resting stream habitats, such as slow, deep pools, when reduction in available habitat by ice in conjunction with reduced water temperature evokes behavioral changes in WCT during winter (Brown and Mackay 1995; Jakober et al. 1998; Brown 1999). This is a

key behavioural change, one that we use to determine habitat suitability within high gradient, mountain headwater streams in Southern Alberta.

Most studies of salmonid overwintering habitat have focused on larger streams and rivers at lower elevations (Cunjak 1996; Huusko 2007). Little detailed work has been done on the identification and location of, as well as site-specific features of, WCT overwintering habitat in high elevation, small, mountain headwater streams. Research in higher elevation headwater streams is hampered by accessibility during winter, as well as by limited available technology to study these environments during winter under different snow and ice conditions.

The research presented here examined the winter ecology of WCT in small headwater streams. Specifically, the research focused on identifying and characterizing physical, chemical, and biological properties of overwintering habitat, as well as the use of such habitat by the resident WCT populations. Stream pools were identified as the primary habitat of interest.

We hypothesized that WCT abundance will increase in identified overwintering pools with the greatest habitat quality, due to the suspected decrease in availability of other habitat, specifically the exclusion by cold water temperatures and the presence of ice. Overwintering habitat with the greatest habitat quality should be those pools comprising of low velocity, increased depth, moderated water temperatures, and sufficient cover.

Methods

Study Area

The study streams are located in the headwaters of the upper Oldman and Crowsnest River watersheds in southern Alberta, Canada. Blairmore Creek, Gold Creek, and Daisy Creek, are three small headwater streams, classified as 5th order (Alberta Environment and Parks 2017). Study sites are located along the three streams at elevations between 1340 m to 1880 m, and with main channel slopes ranging between 19.79 to 22.51 m/km. The study creeks were chosen because of their proximity and similar size, accessibility in both summer and winter, and the presence of near pure or pure WCT (pure being defined as ≥ 0.99 ; Fisheries and Oceans Canada 2014). Other species found in these creeks are Bull Char *Salvelinus confluentus* (Daisy Creek), Brook Char *Salvelinus fontinalis*

(Blairmore Creek and Gold Creek), Mountain Whitefish *Prosopium williamsoni* (Blairmore Creek and Daisy Creek), and Rainbow Trout (Blairmore Creek and Daisy Creek).

Habitat Assessment

All streams were surveyed along their entire length to quantitatively characterize overwintering habitat within the streams' main stems. Each stream was stratified into three segments (canyon, alluvial, headwater), as these segments dictate the number, permanency, formation and type of pools found there. Each stratum occurs in a consistent sequential order on each stream where the canyon stratum is characterized by a highly confined, bed rock dominated channel (lowest elevations, 5th order); the alluvial stratum has an unconfined and alluvium (fluvial or glacial) dominated channel (mid elevations, 4th order); the headwater stratum is characterized by a mix of canyon and alluvial features (highest elevations, 4th-3rd order).

The study focuses on pools. Pools were defined loosely as areas within the stream, with an average lower velocity and greater depth than the surrounding habitat. All pools, throughout each of the streams' entire length were marked with a GPS, characterized by type using a habitat classification system (modified from Flosi et al. 1998), and rated for overwintering habitat quality. This rating (low, medium, high) was based on subjective sampling (Brown and Austen 1996), where measured habitat values and the available literature suggested where fish would be expected to overwinter. Suitable overwintering habitat was rated based on the observed maximum pool depth, water velocity, and available cover. Pools that were very deep, slow flowing, and had cover were classified as high quality pools. Pools that had diminished factors of depth, flow or cover, were classified as medium quality. If pools were shallow, faster flowing, or had little cover, they were classified as low quality pools. A total of 588 pools were identified among the three study streams: 172 on Blairmore, 177 on Daisy, and 253 on Gold. A total of 135 pools (approximately 45 pools per stream or 23 % of the total number of pools) were monitored from summer throughout the winter. Since high quality pools were limited, all high quality pools were sampled. Thirty randomly selected low and medium quality pools were monitored for each stream (5 low and 5 medium quality pools each per stratum).

Additionally, temperature (°C) was measured in each selected pool. During the winter, the fraction of surface ice cover of each pool was visually estimated. In addition, pool dimensions such as wetted width, length, and depth were measured. Pool dimensions were only measured during the summer, because ice conditions did not allow for these measurements in winter with comparable methods. However, these measurements are not expected to differ greatly if taken during base flow. Pool dimensions allowed for estimations of pool area (m²). Pool area was calculated using an ellipse area equation:

$$A = \pi w l$$

Where π = constant; w = wetted width divided by two; l = length divided by two.

Fish Survey

Due to WCT being a threatened species, fish population surveys were performed using the least invasive methods possible. Snorkel surveys were employed in the summer and winter, and were supplemented by camera footage under ice cover. Snorkeling has been used as a noninvasive technique and allows for estimations of population size, size class, and habitat use (Thurrow 1994; O'Neal 2007). Snorkel surveys were used to estimate fish distribution within each study stream, as well as to estimate change of habitat use across seasons when fish begin to congregate in overwintering habitat. Within selected pools, fish were counted, identified to species, and put into size classes by 10 cm increments (< 10 cm < 20 cm, < 30 cm, < 40 cm, < 50 cm), except for <5 cm individuals, as suggested by Thurrow (1994).

Suitable overwintering pools were monitored in the summer and throughout the winter to identify use by WCT. Where snorkel surveys were impossible due to ice cover and low temperatures, a commercial ice-fishing camera was used under the ice (Carlson & Quinn 2005). We used an Aqua-vu® fishing camera (model 760cz), either with an Aqua-vu® ice fishing tripod (Aqua-vu Mo-Pod) when surface ice was present or a modified 4-m-long painting pole when no surface ice was present, but air temperatures were too cold for snorkeling. If a hole through the ice was needed, a 16-cm diameter hand auger was used. A single hole through the ice in the pools center allowed for a 360° view of the pool in real time on the adjacent Aqua-vu® fishing camera monitor.

Fish surveys took place once at the end of the summer, and twice over the winter in all pools. Summer sampling was carried out from August 16, 2016 to September 2, 2016; early winter sampling occurred from November 23, 2016 to December 21, 2016; and mid-winter sampling occurred between January 16, 2017 and March 3, 2017.

Data Analysis

Microsoft Excel (Microsoft, 2016) was used for data management, and the programming language R (R Core Team, 2016) running within RStudio (RStudio Team, 2015) was used for statistical analysis. The library ggplot2 in R was used for graphical display of data (Wickham 2009). Temperature and percent ice cover was compared by streams and sampling periods. Fish abundance was compared by streams, sample periods, strata, habitat, and size class separately. Data were checked for normality using a Shapiro-Wilk test (Quinn and Keough 2002). If after transformations, data remained non-normally distributed, a one-way permutation analysis of variance (permANOVA) was chosen instead, using the library LmPerm in R (Wheeler and Torochiano 2016). The basis of a permANOVA is the comparison of the data distribution among groups to a random draw of those numbers. After a set number of permutations (10,000), the likelihood p will be given, that the real dataset is different from a random shuffle of the data. To make multiple comparison between means, Tukey's (honestly significant differenced or HSD) was used (Quinn and Keough 2002).

Results

Physical Habitat Variables

Water temperatures ranged from 4.22 to 16.70 °C in summer and -0.05 to 3.61 °C in winter. Gold Creek was significantly colder in summer (permANOVA: $F_{2,128} = 25.97, P = 3.41 \cdot 10^{-10}$), and significantly warmer in winter (early winter, permANOVA: $F_{2,133} = 101.2, P = <0.001$; mid-winter, permANOVA: $F_{2,132} = 30.19, P = 1.6 \cdot 10^{-11}$) compared to the other two streams. Blairmore and Daisy creeks only differed significantly in mid-winter (Table 1). Maximum pool depth varied among high, medium and low quality pools among all study creeks (Table 2).

Measurements were taken at base flow levels, where very low water levels were observed on all study

streams. Baseflow conditions continued into the fall and winter. Two reaches on Gold Creek and one reach on Daisy, ranging from approximately 360 m to <50 m, were found to have no surface water (only subsurface flow).

Percent surface ice cover varied among streams and strata (Figure 1). Percent ice cover within the canyon stratum was significantly higher in Blairmore and Daisy creeks during early winter (permANOVA: $F_{2,291} = 471.8$, $P = <0.001$). During mid-winter all three streams were significantly different, with Gold Creek having less stream ice than Blairmore and Daisy creeks (permANOVA: $F_{2,291} = 220.2$, $P = <0.001$). Blairmore Creek had significantly more ice cover than both Daisy and Gold creeks within the alluvial stratum

Table 1. Mean and standard deviation for water temperature (°C) within pools for 3 study creeks across sampling periods during 2016-2017.

Creek	Summer	Early Winter	Mid-Winter
Blairmore	10.37 ±2.55	0.01 ±0.18	0.05 ±0.23
Daisy	9.51 ±1.77	0.27 ±0.61	0.87 ±1.06
Gold	7.49 ±1.61	2.14 ±1.16	1.43 ±0.94

Table 2. Mean and standard deviation for maximum depth (m) of pools rated high, medium, and low quality (2016-2017).

Creek	Quality	Number Sampled (Total number of pools)	Depth (m) (mean±SD)
Blairmore	High	12	0.86±0.24
	Medium	15 (66)	0.74±0.19
	Low	15 (94)	0.54±0.10
Daisy	High	15	0.93±0.46
	Medium	15 (76)	0.82±0.36
	Low	15 (86)	0.58±0.17
Gold	High	17	1.29±0.50
	Medium	15 (109)	0.84±0.19
	Low	15 (127)	0.59±0.15

during early winter (permANOVA: $F_{2,201} = 386.1$, $P = <0.001$). During mid-winter Blairmore Creek continued to support significantly higher percent ice cover (permANOVA: $F_{2,195} = 22.21$, $P = <0.001$). Percent ice cover within the headwater stratum was significantly lower in Gold Creek during early winter (permANOVA: $F_{2,255} = 75.43$, $P = <0.001$). However, by mid-winter, only Blairmore Creek continued to support significantly more surface ice (permANOVA: $F_{2,261} = 15.03$, $P = <0.001$).

WCT Abundance

The total number of WCT observed during sampling periods varied: summer = 2,185; early winter = 994; mid-winter = 870. Total WCT abundance (number of fish/m²) differed among all three study streams and seasons (Table 3).

WCT abundance was significantly higher during summer for Blairmore (permANOVA: $F_{2,120} = 27.61$, $P <0.001$) and Daisy (permANOVA: $F_{2,123} = 8.54$, $P = 0.0003$) creeks; however, no significant difference was found between early winter and mid-winter. WCT abundance was also almost significantly higher in summer for Gold Creek (permANOVA: $F_{2,135} = 3.016$, $P = 0.052$).

WCT abundance was significantly higher during summer in the canyon stratum for Blairmore (permANOVA: $F_{2,51} = 28.05$, $P <0.001$), Daisy (permANOVA: $F_{2,36} = 5.85$, $P = 0.006$), and Gold (permANOVA: $F_{2,51} = 15.12$, $P = <0.001$) creeks. WCT abundance was only significantly higher during summer in the alluvial stratum for Blairmore Creek (permANOVA: $F_{2,27} = 6.018$, $P = 0.007$). WCT abundance was only significantly higher in summer in the headwater stratum for Blairmore Creek (permANOVA: $F_{2,33} = 5.27$, $P = 0.010$) and Daisy Creek (permANOVA: $F_{2,50} = 12.99$, $P <0.001$). There was no significant difference in abundance between early winter and mid-winter for any stratum.

Habitat Occupation

Among all three streams, proportionally WCT abundance was not significantly different among pool classes. WCT abundance was significantly higher during summer in high quality pools for Blairmore Creek (permANOVA: $F_{2,32} = 10.42$, $P = 0.0003$) and Daisy Creek (permANOVA: $F_{2,36} = 3.89$, $P = 0.0296$). Gold Creek did not show significant differences in WCT abundance in high class pools across seasons.

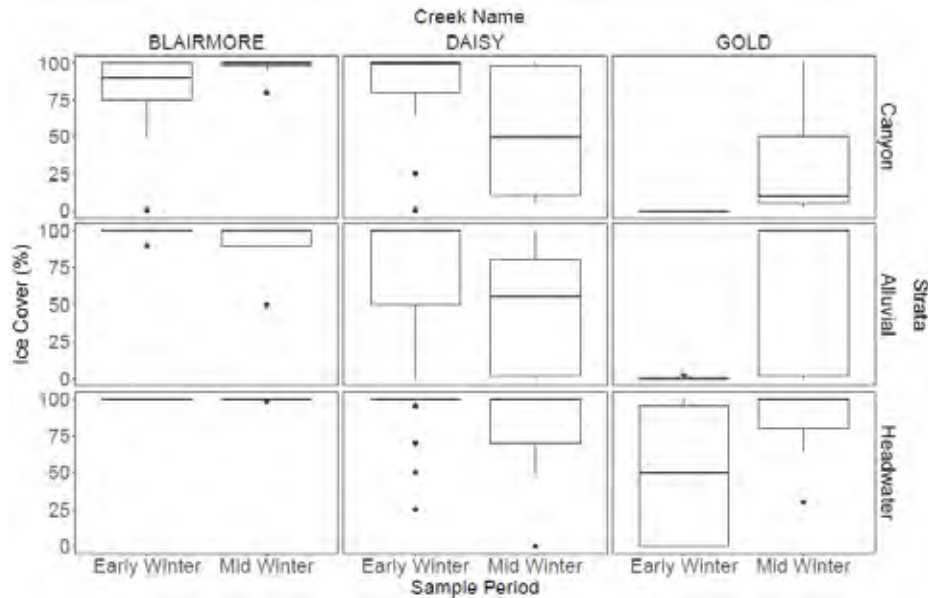


Figure 1. Surface ice cover (%) within overwintering pools, by sample period, stratum and study creek (2016-2017).

Table 3. Mean and standard deviation for WCT abundance (number of fish/m²) within pools for 3 creeks across sampling periods (2016-2017).

Creek	Summer	Early Winter	Mid-Winter
	(mean number of fish/m ² ±SD)		
Blairmore	1.14±0.91	0.33±0.44	0.22±0.28
Daisy	0.67±0.63	0.30±0.45	0.26±0.38
Gold	0.17±0.28	0.07±0.17	0.08±0.18

WCT abundance was significantly higher during summer in medium (permANOVA: $F_{2,40} = 11.11$, $P = 0.0001$) and low (permANOVA: $F_{2,42} = 6.93$, $P = 0.0025$) quality pools for Blairmore Creek. No significant difference was detected for WCT abundance for Daisy and Gold creeks for both medium and low quality pools. There was no significant difference in abundance between early winter and mid-winter.

Size Classes

WCT size class abundance varied among sample periods and creeks (Figure 2).

Abundance for size class < 5 cm was significantly higher during summer among sampling periods for

Blairmore (permANOVA: $F_{2,120} = 16.57$, $P < 0.001$), Daisy (permANOVA: $F_{2,123} = 9.44$, $P = 0.0002$), and Gold (permANOVA: $F_{2,138} = 3.44$, $P = 0.0347$) creeks. WCT abundance for size class < 10 cm was significantly higher during summer for Blairmore (permANOVA: $F_{2,120} = 24.82$, $P < 0.001$) and Daisy (permANOVA: $F_{2,123} = 8.22$, $P < 0.001$) creeks. Changes in WCT abundance for size class < 10 cm were not significant for Gold Creek. WCT abundance for size class < 20 cm was significantly higher during summer for Blairmore (permANOVA: $F_{2,120} = 4.564$, $P = 0.0123$) and Gold (permANOVA: $F_{2,138} = 3.63$, $P = 0.029$) creeks. Change in WCT abundance for size class < 20 cm in Daisy was not significant. Lastly,

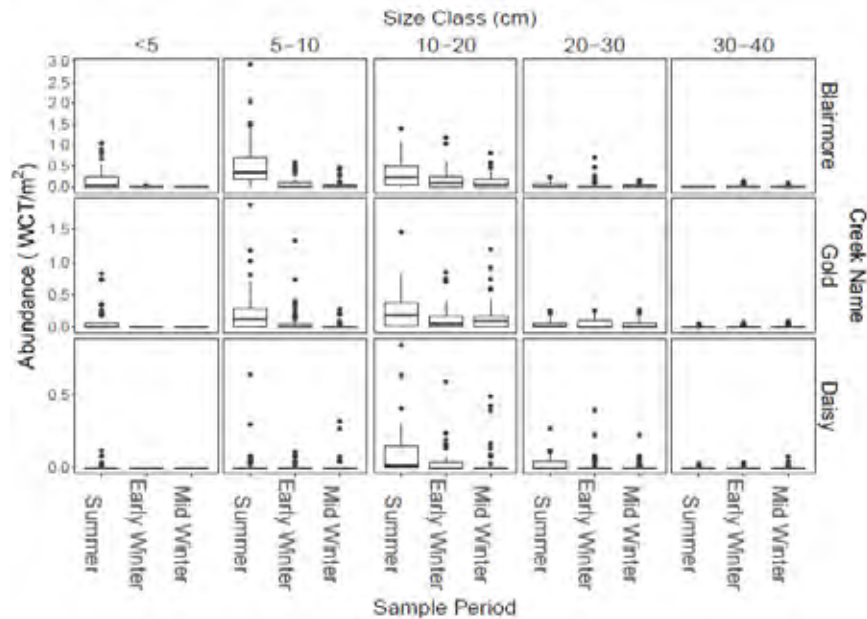


Figure 2. Changes in Westslope Cutthroat trout abundance (WCT/m²) per size class (cm), sample period, and study creek within overwintering pools (2016-2017).

there was no significant difference in abundance between early winter and mid-winter for any of the smaller size class, across all streams.

WCT abundance for larger size classes < 30 cm, <40 cm was not significantly different among sampling periods for any of the study streams. However, when looking more closely at specific strata, a significant decrease was detected for < 30 cm abundance on Gold (perMANOVA: $F_{2,48} = 972.6$, $P = < 2^{-16}$) Creek, but no significant difference was detected in the canyon strata between Blairmore and Daisy creeks. No WCT were classified in the < 50 cm size class during this research for any of the study creeks.

All size classes present during winter also appeared to be feeding during our surveys. Although our Aqua-vu[®] camera did not record video directly, we were able to capture video via a digital camera on the LCD screen, as well as during snorkeling surveys. Opportunistic drift feeding was mainly observed, occurring at temperatures as low as -0.06 C°.

Discussion

Overwintering pools for WCT in the study streams varied in habitat quality and ice cover stability. Temperature and surface ice cover were identified as the main drivers of pool occupation by WCT.

Gold Creek was the most moderated study stream, with colder temperatures in the summer and warmer temperatures during winter sampling compared to the other two streams. As a result, percent surface ice cover was significantly less in Gold Creek than both Blairmore Creek and Daisy Creek throughout winter. Seasonality determined WCT abundance among streams and size classes, and an overall decrease in WCT abundance was observed. Decreased abundance occurred across all streams, with the largest decline of fish numbers in the smaller size classes. No difference in abundance for larger specific size classes were found among sampling periods. However, there was a significant decrease in abundance within canyon stratum of Gold Creek.

Differences in the thermal regime were observed among the study streams. Especially in Gold Creek, water temperatures were moderated over the seasons. Temperatures in Gold Creek remained lower than the other two streams in the summer, but higher in the winter. The moderated temperatures in Gold Creek indicate a strong influence of groundwater to the creek. High influx of groundwater can significantly shape fish habitat and modify habitat use in winter (Power et al. 1999). In temperate streams, groundwater influx has a significant influence on overwintering salmonids

by reducing ice cover and hence altering habitat quality (Brown and Mackay 1995; Brown 1999). In contrast, Blairmore Creek did not have groundwater-moderated temperatures and thus sustained far greater ice cover.

Consistent with Brown and Mackay (1995) and Brown (1999), we observed WCT aggregating in predominantly high quality pools. However, we observed a general decrease in WCT abundance for all streams, pool qualities, and strata from summer to winter, even in large, high quality pools. Although, total WCT abundance did not differ significantly between early and mid-winter, likely because the habitat conditions remained stable during these periods.

The decrease in WCT abundance between summer and winter was most pronounced in small size classes. Smaller size classes were expected to decline over winter as young of the year and juvenile salmonids seek refuge in substrate during winter (Cunjak 1988; Heggenes et al. 1993). Blairmore Creek, which had the largest proportion of smaller size classes, showed the largest decrease in WCT abundance from summer to winter.

Larger WCT size class abundance did not differ significantly among sampling periods in general. Summer habitat was likely influenced by base flow water levels (sustained only by deep groundwater storage and shallow subsurface flow), which continued into the fall and winter (personal observations). During late summer, stream discharge was very low, especially for headwater and alluvial portions of these streams, where total loss of surface flow in some reaches was observed. This reduced connectivity might lead to high fish abundance in pools during the summer which will carry over to less movement and smaller home ranges of adult trout throughout the seasons.

WCT abundance for larger size classes (< 30 cm - < 40 cm) in Blairmore Creek and Daisy Creek did not differ significantly between winter sampling periods. These streams, however, experienced the greatest decrease in water temperature during the winter, typically remaining just above freezing (on average 0 C°). Very low water temperatures lower fishes' ability for swimming (Bernatchez and Dodson 1985; Heggenes and Traaen 1988) and reduce feeding (Cunjak and Power 1986). Reduced activity is most likely occurring in WCT populations in Blairmore and Daisy creeks due to the low water temperatures observed. In addition, these temperatures allowed

most of the stream to maintain snow and ice cover throughout the winter. Surface ice is a significant factor as cover in winter, which can affect fish activity (Linnansaari et al. 2008) and use of habitat (Meyer and Griffith 1997). Meyer and Griffith (1997) found a substantial increase in young-of-the-year Rainbow Trout density when surface ice was present in habitat enclosures, indicating its importance. When pools are no longer covered with surface ice, this may increase the chances of predation and limit its use during winter if no other cover is present. Complete surface ice and snow created stable pool conditions which allowed for WCT to reduce energy depletion and predation risk. In these streams, when surface ice has finally set and snow begins to accumulate, the water's contact with air temperatures is limited. Snow and ice insulates, preventing super-cooling of water and thus further development of frazil and anchor ice conditions.

Distribution of larger WCT within a stream varied substantially by stream strata. While adult WCT abundance (< 30 - < 40 cm) was maintained across seasons in the alluvial and headwater strata of Gold Creek, the canyon strata lost nearly 89.6 % of their population in high quality pools throughout winter. These larger size classes were suspected to be too large to conceal in cover interstices within the substrate. Thus, large size classes were also thought to be subject to the habitat limitation idea, which is the 'squeezing effect' from cold water temperatures, ice and limited pool habitat (Heggenes et al. 1993; Brown and Mackay 1995; Brown 1999; Jakober et al. 2000). Very little subsurface ice development and surface ice persistence was observed in this Gold Creek stratum due to higher water temperatures. WCT in these heavily groundwater influenced areas may take advantage of higher winter water temperatures and may move to other, more productive habitats, such as riffles and runs, in search of food during winter. Food may become more important to fish in these warmer conditions, as increased water temperature increases the rate of metabolism, and thus increases energy demands (Cunjak 1988b). Cunjak (1988b) found depletion of metabolic reserves occurred during stable warm winter conditions for immature Brook Trout and Brown Trout *Salmo trutta*, when daily mean water temperatures were between 2° and 5° C within spring-fed tributaries. Similar temperatures were observed in certain pools within Gold Creek and are attributed to ground water input. For example, one pool had water

temperature measured at 2.72 C in mid-winter and 3.78° C in early winter. Additionally, WCT and aquatic macroinvertebrates were observed in riffles and runs during the early and mid-winter surveys on Gold Creek, suggesting, that WCT maintain active feeding over the winter, when waters are warmer.

Conclusion

Overwintering habitat is often assessed from fish aggregations during fall and spring surveys, and by assessing the quantity and extent of deep, low-velocity pool habitat. It is assumed that not only do large numbers of fish move to these areas to overwinter, but also that fish must continue to stay in these areas throughout winter to avoid deleterious ice conditions and extreme coldwater temperatures. While this was previously known for larger streams and rivers, we were able to observe the same pattern for small, mountain headwater streams in Alberta, Canada. Two types of stable pool habitat were found in our study streams during winter: those which continuously sustain cold water temperatures and surface ice cover, and those that continue to stay free of surface ice while maintaining warmer water temperatures. Both stable habitats provide shelter from anchor and frazil ice formation within large areas of the stream.

We confirmed, that overwintering pools are the most important habitat for adult WCT survival during winter in high gradient mountain streams, especially when unstable ice conditions exclude other habitats. We were able to observe a large fraction of adult WCT of the summer population in the winter in the same pools. We suggest further investigation by methods that would compliment snorkel surveys and under water cameras, such as radio telemetry, in order to quantify movement patterns rather than simple occupation of specific habitats.

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Using Statewide Survey Data to Support Local-Scale Management of Michigan Trout Streams

Troy Zorn¹, Todd Wills and Jan-Michael Hessenauer², Ed Bissell and Joel Lenz³, Ashley DePottet⁴, Danielle Forsyth Kilijanczyk⁵, Anila Francis⁶

¹Marquette Fisheries Research Station, Michigan Department of Natural Resources, 484 Cherry Creek Road, Marquette, MI 49855

²Lake St. Clair Fisheries Research Station, Michigan Department of Natural Resources, 33135 South River Road, Harrison Township, MI 48317

³Department of Geography, Environment, and Spatial Sciences- Remote Sensing and Geographic Information Systems, Michigan State University, 673 Auditorium Road, 2nd Geography Building, East Lansing, MI 48824-1117

⁴380 New York Street, Redlands, CA 92373

⁵Institute for Fisheries Research, Michigan Department of Natural Resources, 400 North Ingalls, G250, Ann Arbor, MI 48109

⁶Michigan Department of Technology Management and Budget, Romney Building - Tenth Floor, 111 S. Capitol Ave, Lansing MI 48933

Abstract—Building upon insights from state fishery managers and multi-decadal studies on trout populations, the state of Michigan embarked on a statewide program to monitor trends in wild trout populations and to assess the status of fish assemblages and habitats in all streams. Initiated in 2002, the Stream Status and Trends Program (SSTP) provides a rich set of information from standardized surveys across Michigan. Data from SSTP surveys support public-facing tools which inform stakeholders (biologists, anglers, various publics) of trends in highly-valued trout populations and provide an empirical basis for managing individual stream reaches. Launched in 2014, the Stream Fish Population Trend Viewer allows users to assess trends in wild trout abundance, growth, and survival using data from fixed (index) electrofishing sites throughout the state. The Michigan Stream Evaluator, scheduled for release in 2017, provides comparisons of habitat and fish assemblage conditions at individual sites to benchmarks computed from surveys in streams having similar size, temperature, channel gradient, and geographic location attributes. Both tools are refreshed annually with additional survey data.

Introduction

Conserving and managing wild trout populations and fisheries, and aquatic communities in streams in general, pose challenges on several levels to fisheries biologists. Trout populations are dynamic through time, being shaped by short- and long-term changes in flow, temperature, and water quality conditions (Lobón-Cerviá 2004; Zorn and Nuhfer 2007a). Longer-term climatic and hydrologic changes further complicate patterns in trout recruitment, growth, survival, and ultimately, population interactions and trajectories (Poff and Ward 1989; Wenger et al. 2011). Even for hydrologically stable, groundwater-fed streams, 10-20 years of annual population estimate data are needed just to characterize baseline variability in trout populations (Wiley et al. 1997).

Trout populations live in a diverse array of habitats, ranging from high mountain lakes and streams, to low-gradient groundwater-fed rivers

(Frissell et al. 1986). Even among groundwater-influenced streams in the Great Lakes region, flow stability, reach-scale stream gradient and power can vary over several orders of magnitude, resulting in myriad combinations of stream habitat and fish assemblages across the region (Zorn et al. 2002). In addition, the streams in which trout reside are constantly changing as stream channel habitats are shaped and altered by natural and human-induced geomorphic processes (Ward and Stanford 1983; Poff et al. 1997).

In water-rich Midwestern states like Michigan, there are tens of thousands of kilometers of trout stream habitat. Wild trout production from these streams dwarfs that of state fish hatcheries, making these streams a tremendously valuable resource. Demands on Michigan's fishery managers are diverse though, with upwards of 180,000 stream trout anglers, and major fisheries in the Great Lakes and over 10,000

inland lakes. With so much trout water and so many fisheries to protect and manage, much of the challenge for biologists is to make sense of stream survey data for an individual water body (or location) that may be sampled only once every couple decades. Some basic questions need to be answered. For example, to what degree do conditions at the site, reach, and catchment scales influence what the survey data show? How is the observed population shaped by regional climatic or hydrologic processes that similarly affect populations across the region? What are typical stream habitat, fish assemblage, and fish population characteristics for this type of stream? Are conditions out of the ordinary, or in need of specific management action? Answering these questions requires a longer-term, regional perspective that goes beyond typical management unit or political boundaries, and necessitates sampling and analyses coordinated across larger spatial and temporal scales. An improved inventory program was needed so trout managers (with limited local-scale data) could better understand and explain what was driving fish populations at individual stream sites.

Michigan's Stream Status and Trends Program (SSTP; Hayes et al. 2003) was initiated in 2002 to characterize differences among a diverse array of stream systems and to describe trends in key fish populations over time. Ultimately, the intent was to be able to provide the information needed to address questions relating to spatial and temporal variation in stream fish (especially wild trout) populations throughout the state. The SSTP brought the accompanying challenge of conveying the importance of temporal and spatial influences on stream fish populations to interest groups and a public typically less exposed to this broader perspective. Here, we show how data from the SSTP are packaged and delivered to address questions that anglers, the public, and other biologists often ask local fishery managers.

Methods

The SSTP grew from recognition of existing knowledge gaps and understanding acquired from earlier stream fish ecology studies in Michigan. Analyses of long-term index site data on Brook Trout *Salvelinus fontinalis* and Brown Trout *Salmo trutta* populations in the Au Sable River and Michigan waters highlighted the importance of long-term population index data to trout ecology and management (e.g., Clark et al. 1980; Zorn and Nuhfer 2007a; 2007b).

An appreciation of the diversity of Michigan streams and the need to better understand stream-specific influences on fish populations was evident from several studies completed in Michigan during the last two decades (e.g., Wiley and Seelbach 1997; Wiley et al. 1997; Zorn et al. 2002; Wehrly et al. 2003; Seelbach et al. 2006). Combined, these efforts highlighted a fundamental need for understanding what was driving spatial and temporal patterns in fish populations and habitat conditions in Michigan streams.

The design of the SSTP incorporates two different, yet complementary, types of sampling. Fixed (index) site sampling is used for stream types supporting valuable fisheries for wild trout or Smallmouth Bass *Micropterus dolomieu* streams. Wadeable electrofishing reaches (typically 1,000 ft) are sampled in late-summer using a 3-years-on, 3-years-off rotation, to provide broader geographic coverage throughout Michigan while enabling estimates of year-to-year survival of trout at individual sites. The following parameters are measured during years when sampling is scheduled: mark-recapture population estimates by size and age group of trout (1-pass catch rates for Smallmouth Bass); annual estimates of trout growth and survival; and hourly water temperature measurements. Instream, riparian, and woody habitat conditions and fish community composition of electrofishing reaches are measured once per 3-year-on cycle to enable assessment of effects of river- and site-level attributes on fish populations (Wills et al. 2008). Overall, this approach provides high-resolution trend detection, with the regional network of sites providing information on the spatial extent of trends and synchrony among populations (Zorn and Nuhfer 2007).

A stratified random sampling design is used primarily for general resource inventory, with the intent of quantifying fish assemblage and habitat conditions in each type of stream in Michigan. The primary sampling unit for the stratified random sampling design is the river valley segment (Seelbach et al. 2006), a contiguous segment of a stream that is characterized by similar hydrology, water quality, channel morphology, riparian land cover, and fish communities along its length. For the SSTP, individual valley segments were randomly ordered for sampling by field crews. We expect that it will take several decades before the entire list of segments has been sampled. Surveys at random sites involved sampling the fish assemblage in the sample reach via

single-pass electrofishing, hourly water temperature measurements, and collection of data describing instream, riparian, and woody habitat conditions within the sample reach (Wills et al. 2008). Data from all SSTP surveys are entered into the Michigan Department of Natural Resources (MDNR) Fisheries Division's centralized database, the Fish Collection System (FCS). Seeing the value of the standardized sampling approach and thorough data collection procedures, MDNR Fisheries Division staff and other partners have completed many additional surveys using SSTP random site protocols, providing additional survey data for use.

The SSTP initiated use of standardized data collection protocols for stream surveys across the state and centralized storage of survey information. After the SSTP was initiated, several years were needed for data to accumulate in the FCS to reach a “critical

mass” for use in development of decision support tools. In addition, a modest amount of external funding had to be acquired to support the additional expertise needed for tool development.

Stream Fish Population Trend Viewer (TV)- Stream survey data for Brook Trout, Brown Trout, Rainbow Trout *Oncorhynchus mykiss*, Coho Salmon *Oncorhynchus kisutch*, and Smallmouth Bass at fixed sites are queried to provide annual estimates of total biomass density, numerical density (numbers per acre by age and size class), and mean length-at-age and annual survival for age classes with adequate sample sizes. These data are annually extracted and summarized from the FCS for use in the TV via MS-Access ODBC queries. Average values for each parameter at that site are also calculated from surveys since 2002, and represent “long-term” mean values for that parameter at that site. These data allow users

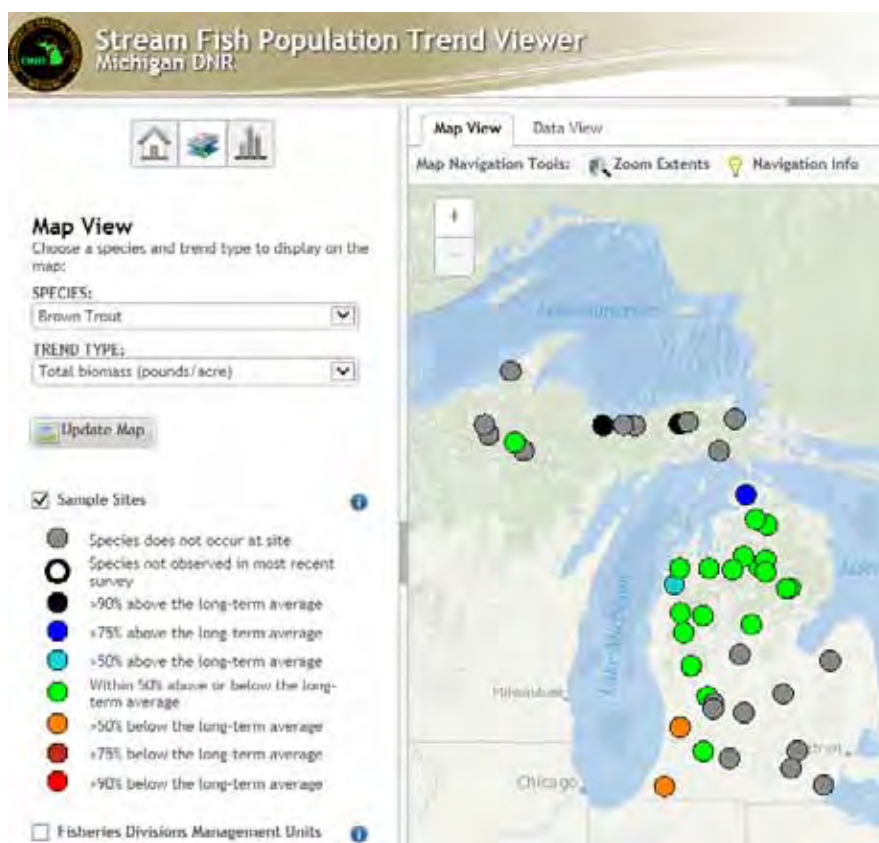


Figure 1. Screen capture from the Stream Fish Population Trend Viewer showing total biomass (pounds/acre) of Brown Trout from most recent surveys at electrofishing index (SSTP fixed) sites in Michigan. Color of dots shows how most recent survey value compares to average value at that site from surveys conducted since 2002. Accessed May 9, 2017.

of the TV to view current status (percent departure from “long-term” average) for a given parameter (e.g., density of age-1 Brown Trout) at all sites in Michigan on a map, and assess the extent to which other populations in the region share a similar status (i.e., the spatial extent of the trend). Data for each parameter measured at fixed sites are also viewable in graph and tabular form, and downloadable as .pdf or .xls files. Historic (pre-2002) data are also available at several fixed sites.

Michigan Stream Evaluator (MSE)- The MSE enables users to access benchmark or typical values for many parameters measured on hundreds of surveys conducted using MDNR SSTP random site protocols. Benchmark values (mean and standard deviation) of various survey parameters are computed via MS-Access ODBC queries of data in the FCS. Survey parameters include numerical density of fishes, density by size class for game species, transect-based instream habitat data, bank and riparian habitat measures, density of logs and woody habitat, and others. The

MSE includes benchmarks for over 100 survey parameters. Data are stratified in various ways so that benchmarks are computed for the diversity of stream types and regions of Michigan. Strata are based on river catchment size, July mean water temperature, reach gradient, geographic area or region, and accessibility to the Great Lakes.

Users of the MSE can select an individual stream survey, the attributes of benchmark for comparisons. Values of parameters from the chosen survey are displayed, along with mean and standard deviation values computed from benchmark streams. For each parameter, a line graph depicts the difference between observed and benchmark mean values, with the difference standardized (divided) by the benchmark’s standard deviation.

The Michigan Department of Technology Management and Budget (DTMB) hosts the TV and MSE on secure Windows-based Servers running Internet Information Services. Microsoft SQL Server, enabled with ESRI ArcGIS Spatial Database Engine houses the GIS and tabular data for both tools. ESRI ArcGIS for Server provides the technology to serve the mapping data. ESRI JavaScript application programming interface powers the mapping and analysis in the TV and MSE. JavaScript charting libraries enable the interactive visualization of fish abundance and habitat specific information. Data for the TV and MSE are updated annually (typically in late winter) after fish sampling and age and growth data from the previous field season have been entered and approved.

Results

Through the TV, the most recent data documenting trends in wild trout abundance, growth, and survival are efficiently summarized and made publicly available in a relatively user-friendly form. Data are available for fixed sites on 16 wild trout streams with Great Lakes access, 19 wild trout streams without Great Lakes access, and 9 Smallmouth Bass streams, with surveys at some locations going back as far as 1947. A link to the TV can be accessed by entering “Stream fish population trend viewer” into an internet search engine.

Map-based outputs of the TV make it easy to assess population trends at a site, and to compare conditions among sites. For example, recent data on total Brown Trout biomass from fixed sites around

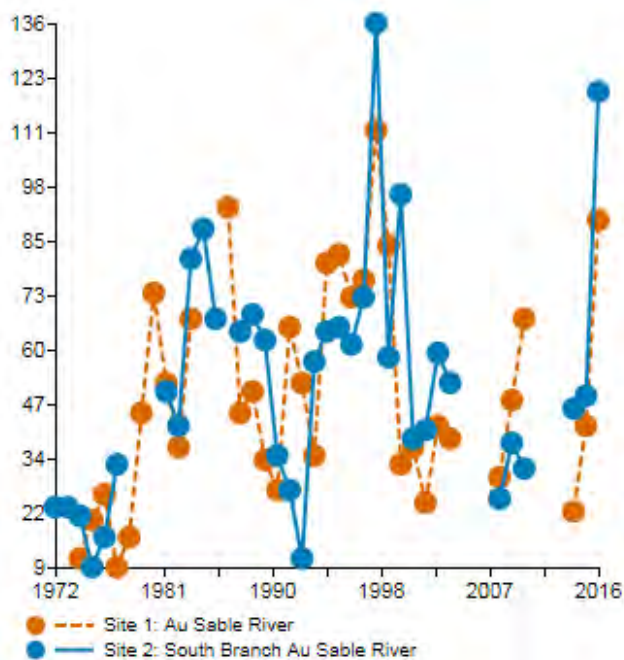


Figure 2. Screen capture from Stream Fish Population Trend Viewer showing numerical density (number/acre on y-axis) of age-1 Brook Trout from surveys since 1972 at fixed sites on the main-stem Au Sable River and South Branch Au Sable River. Accessed May 9, 2017.

Michigan suggests that among trout streams where Brown Trout occur, biomass is generally near or above average in streams in the northern portion of the state, though more than 50% below average in some southwest Michigan trout streams (Figure 1).

Tabular outputs from the TV allow users to explore long-term trends in abundance, growth, and survival of wild trout, and assess similarity in populations within a region. For example, long-term data from the main-stem Au Sable River and South Branch Au Sable River show similar patterns in density of age-1 Brook Trout, suggesting population dynamics may be influenced by larger-scale processes acting on trout populations across the region (Figure 2).

Example output from the MSE demonstrates its utility for placing survey results into a broader context. Here, results from a survey on the West Branch Sturgeon River were compared to benchmark values computed from 12 surveys conducted on similar streams in Michigan's Northern Lower Peninsula (Figure 3). The West Branch Sturgeon River was considerably wider than most surveys in its class, and had a notably greater percent coverage of rooted aquatic plants. Woody habitat, particularly the density of large logs, was somewhat lower than average. Total fish and small Brown Trout density were slightly higher than benchmark values, while density of all salmonids larger than 7 in was below average.

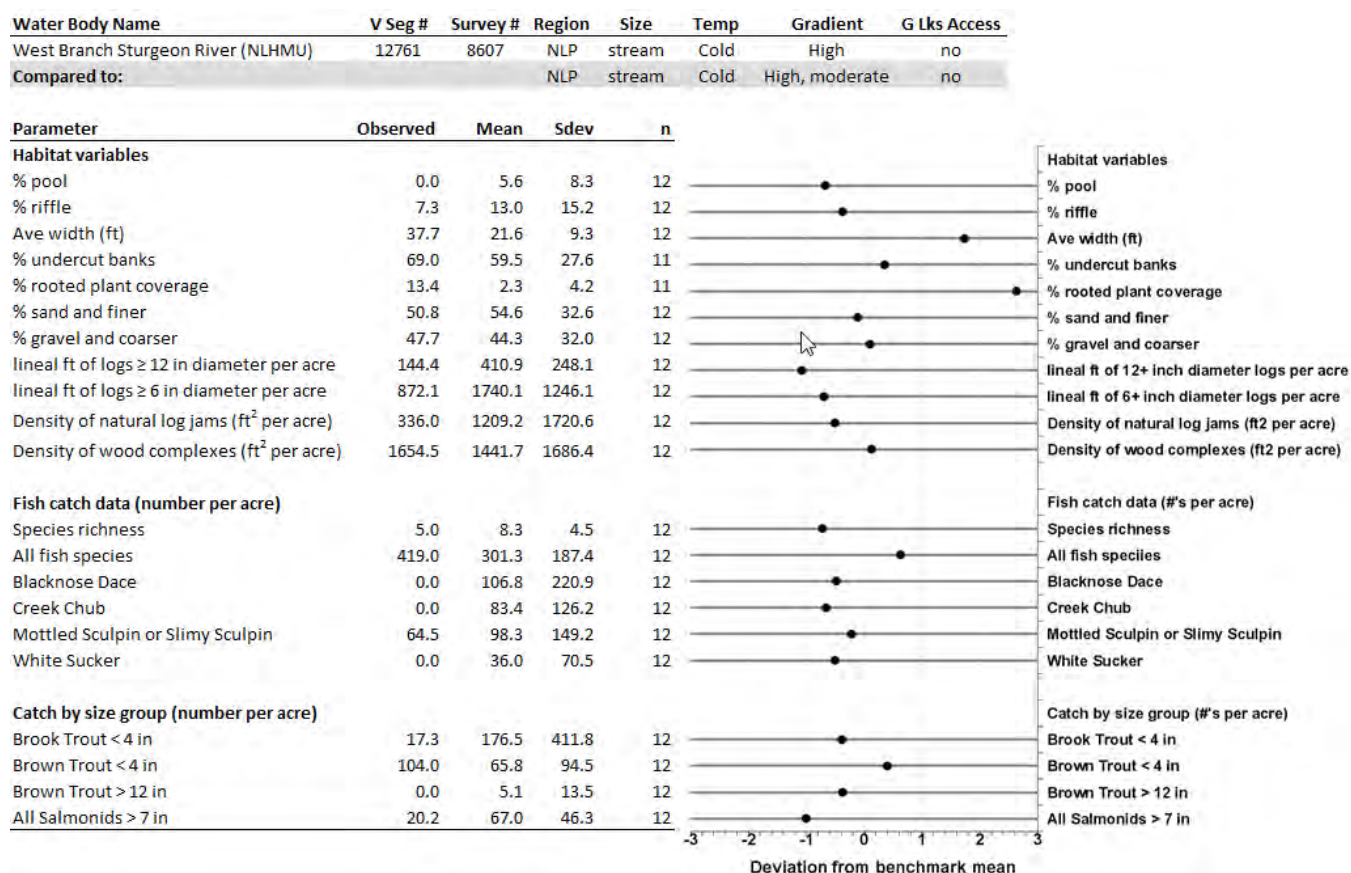


Figure 3. Preliminary graphical and tabular outputs from Michigan Stream Evaluator showing observed fish and habitat sure measurements from a survey on the West Branch Sturgeon River in the Northern Lake Huron Management Unit (NLHMU), compared to benchmark mean and standard deviation (S Dev) values computed from twelve stream segments in Michigan's Northern Lower Peninsula (NLP) having similar size, July mean temperature (Temp), channel gradient, and Great Lakes accessibility (G Lks Access) attributes.

Discussion

The TV and MSE tools provide an efficient means to summarize and present data collected across a broad region in an ecologically meaningful way that benefits local fishery managers, managers of river habitats, anglers, and the public. The tools provide relatively simple diagnostic information, though they represent the culmination of years of technical work, including landscape-scale classification efforts, long-term population dynamics studies, statewide coordinated sampling efforts, over 10 years of coordinated sampling efforts, database management and queries, and integration of databases, GIS technology, and website development. Relatively simple tools built upon technical programs fit well with the diverse set of decision support tool users including agency, university, and tribal biologists; tech-savvy trout anglers; aquatic non-profits; citizen scientists; and interested publics. We anticipate these tools will greatly aid the MDNR in its efforts to openly share science and technical data, and to foster collaboration with others, a goal recently identified in Michigan's statewide Inland Trout Management Plan (Zorn et al. *Under review*).

The TV supports wild trout management, research, and angler outreach by placing the latest survey data directly in the users' hands. The data delivery interface, modeled after the USGS's state-level real-time daily streamflow webpages (e.g., <https://waterdata.usgs.gov/mi/nwis/rt>), provides the latest trout trend information for Michigan in a simple, user-friendly manner. The parameters available (abundance, growth, and survival) are key drivers of populations and important for evaluating effects of environmental or management changes, particularly those associated with flow, water quality, or sport fishing regulations. Since population trends for a given stream reach often are consistent upstream and downstream of survey locations (Wills et al. 2008), and often in nearby waters (Zorn and Nuhfer 2007b), the TV can help managers to better interpret an individual survey, since it will show whether populations in the region at that time are trending high, low, or average.

Data are provided for many of Michigan's most popular trout streams, and therefore are of great interest to fishery managers, tackle shops, anglers, and guides. The regional perspective of the TV provides all with a broader view of regional trends in

trout populations, thus heightening the awareness of climatic and other larger-scale processes that drive trout population dynamics in the state (Zorn and Nuhfer 2007b). Data on riverine Smallmouth Bass populations also supports an improved understanding of dynamics of this popular sport fish. Furthermore, the ability to export or download data at individual sites enables users to satisfy their curiosity through more detailed exploration of trends in fish abundance, growth, and survival.

The easy-to-use MSE represents a breakthrough for biologists managing streams because it provides geographically-relevant, empirical benchmarks for comparison with individual survey results. The strata used in computing benchmarks (size, temperature, gradient, region) represent key large-scale factors that drive spatial variation in local stream habitat conditions and fish assemblages in Michigan and elsewhere (Zorn et al. 2004; Zorn and Wiley 2006; Steen et al. 2008). As a tool focusing on entire fish assemblages as well as habitat, the MSE provides a basis for examining relationships between stream habitat characteristics (summary strata and field-measured variables) and the distribution and abundance of many fish species, including invasive species.

The MSE is especially useful as a means of characterizing expectations for a stream reach, and flagging measured parameters that exceed expectations (positively or negatively) and may deserve further inquiry. It can be used to address management questions related to fish populations (e.g., Are juvenile Brook Trout densities unusually low at the site?) or fish habitat (Is the reach unusually sandy or lacking in large woody habitat?). By computing benchmarks from survey data for similar river segments, the MSE can provide useful characterizations of expected fish communities and habitats for river segments where surveys do not exist or recent surveys are unavailable. This information can prove especially useful to managers when they need to assess damage to habitats or fish populations (e.g., fish kills) and pre-impact survey data are limited or unavailable.

We expect the TV and MSE tools to remain durable and robust over time. Both tools will be updated annually via queries and maintained on state servers. Trout population datasets for some sites in the TV are among the longest-running trout population data sets in the country, and increased accessibility to

the data will only increase their utility. The hundreds of surveys in the MSE may be somewhat limiting, given the diversity of stream types in Michigan, but this information base will only improve over time, as SSTP random site surveys accumulate and managers make increased use of SSTP random site sampling protocols on discretionary surveys. Strata used in the MSE should be useful over the long-term since they represent foundational landscape factors that structure stream habitat and explain spatial variation in habitat and fish assemblage characteristics in streams across the region (Seelbach et al. 2006; Zorn and Wiley 2006). Data used in these projects can be refreshed efficiently by a skilled biologist and data analyst, and efficiency of data refreshes should increase over time with increased use of standardized methods, improved data entry forms, and database updates.

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Session 5

Native Trout Conservation





Lake Trout Where you Need them—Restoring Reproducing Lake Trout in Michigan Waters of Lake Huron

James E. Johnson¹, Ji X. He

Michigan Department of Natural Resources, Alpena Great Lakes Fisheries Station, 160 East Fletcher, Alpena, Michigan, 49707, USA, ¹Corresponding author: wejay@charter.net; retired

Abstract—Lake Trout *Salvelinus namaycush*, the native deepwater, predator of the upper Great Lakes, was nearly extirpated from Lake Huron in the 1940s. Restoration efforts constitute one of North American's largest-scaled keystone predator recovery projects. From 1973 to 2016 more than 55 million yearling-equivalent Lake Trout were stocked in the main basin of Lake Huron. Until 2000, excessive Sea Lamprey *Petromyzon marinus* numbers and fishing rates killed too many Lake Trout before they reached spawning age. Until 2004, Alewives *Alosa pseudoharengus* dominated diets of Lake Trout, leading to low egg thiamine levels, which reduced viability of Lake Trout fry. Furthermore, adult Alewives fed on Lake Trout fry. By 2004, Alewife had collapsed. While the decline in Alewife was probably essential, a combination of factors contributed to widespread reproduction of Lake Trout since 2003. Beginning in 2000, more effective Sea Lamprey and fishing controls were implemented. Seneca strain Lake Trout survived to spawning age better than Lake Superior and other Upper Great Lakes strains, and therefore became the principal strain stocked after about 1995. Lake Trout management units were enlarged in the western main basin, affording protection from overharvest across feeding and spawning habitats and migratory corridors. Relief of these impediments to reproduction led to improved prospects for rehabilitation of Lake Trout in western Lake Huron.

Lake Trout *Salvelinus namaycush* was the top native offshore predator of Lake Huron (He et al. 2015) and supported valuable commercial fisheries from the 1830s to the 1940s (Baldwin et al. 2002; Brenden et al. 2013). By the late 1940s, Lake Trout had collapsed (Brenden et al. 2013), succumbing to a combination of overfishing and depredation by Sea Lamprey *Petromyzon marinus* (Eshenroder et al. 1992; Eshenroder et al. 1995), and were probably extirpated from the main basin of Lake Huron; only two remnant stocks have been documented, both in Georgian Bay (Reid et al. 2001; Johnson et al. 2004). The Sea Lamprey, which was among many invasive aquatic species to colonize the Great Lakes (Mills et al. 1993; Ricciardi 2001), reached Lake Huron in the 1930s, and parasitized Lake Trout and other species (Hile 1949; Eshenroder 1992; Eshenroder et al. 1995). The Alewife *Alosa pseudoharengus* colonized Lake Huron and became a dominant prey species after Lake Trout had collapsed (Smith 1970). Alewife abundance during the 1950s to 2004 was considered an impediment to Lake Trout reproduction (Fisher et al. 1996; Fitzsimons and Brown 1998; Brown et al. 2005). Alewife collapsed in 2004 (Riley et al. 2008).

Efforts to restore Lake Trout in the Great Lakes constitute one of North American's largest-scaled keystone predator recovery projects (Johnson et al. 2015). Lake Huron alone, for example, measuring 5,666,000 ha in surface area, is approximately five times larger than Yellowstone National Park, where keystone predator recovery is also a priority (Fritts et al. 1997). In the late 1950s, the Great Lakes Fishery Commission implemented a Sea Lamprey control program for the Great Lakes (Siefkes et al. 2013). Restocking of Lake Trout in Lake Huron began in the 1970s (Eshenroder et al. 1995; Whelan and Johnson 2004; Johnson et al. 2015). In 1981-1982, the first spawning and fry production by hatchery-origin Lake Trout were observed on a small inshore reef in western Lake Huron (Nester and Poe 1984). Beginning in 1984, reproduction was documented in Thunder Bay (Figure 1), western Lake Huron; however, after 1990, reproduction appeared to be in decline (Johnson and VanAmberg 1995). Parry Sound (Figure 1), in Georgian Bay, Ontario, is one of only two Lake Huron sites where a remnant stock of Lake Trout is known to have survived the collapse. A self-sustaining Lake Trout population has been reestablished there (Reid

et al. 2001). Persistent Sea Lamprey depredation and localized overfishing contributed to delays in restoring spawning populations of Lake Trout in the main basin, but resolution of these issues during the late 1990s–2000 allowed rebuilding of spawning-stock biomass to targeted levels in all Michigan management units by 2002. These Lake Trout populations, however, were almost entirely composed of hatchery-origin fish (Johnson et al. 2004). Johnson et al. (2004) reported that, in 2003, substantial spawning stocks had been established by stocking of Lake Trout, but reproduction was still far below levels necessary to sustain the species.

This represents the fourth presentation to a Wild Trout Symposium on the Great Lakes Lake Trout restoration program. Joe Kutkuhn (1980) reported at Wild Trout II that 20 years of stocking had produced no signs of reproduction and that extractions (harvest rates) were too high to permit accumulation of spawning stock. Carlos Fetterolf (1984) followed in Wild Trout III with description of interagency organizational approach to rehabilitation and evidence of success in Lake Superior but reported that rehabilitation was still a struggle due to high total mortality rates. In Wild Trout IV, Randy Eschenroder (1989) presented another lament and this time added the stocking of nonnative salmonids to the litany of issues holding back progress. Since Wild Trout IV there have been no updates from the Great Lakes Lake Trout project.

Our purpose in this paper is to use stock assessment data and population modeling to assess whether Lake Trout are now becoming self-sustaining. We used catch-at-age modeling to examine how rehabilitation strategies contributed to the observed progress in reaching rehabilitation milestones, focusing on the following questions: (1) are combined Sea Lamprey- and fishing-induced mortality rates below target levels set by the agencies; (2) is biomass of older and mature fish increasing and/or sufficient to allow for natural reproduction; and (3) does natural reproduction approximate the level of recruitment achieved from stocking and is therefore sufficient to sustain the population? Rehabilitation is likely to succeed when the three criteria are met. Finally, we discuss factors that were crucial to the measured outcomes.

Here, we define the main basin of Lake Huron to be waters of Lake Huron exclusive of the North Channel, and Georgian Bay (Figure 1).

Methods

Stocking

Stocking data were obtained from FWS/GLFC (2017). Most Lake Trout stocked in Michigan waters of Lake Huron were raised by the U. S. Fish and Wildlife Service Jordan River, Iron River, and Pendills Creek National Fish hatcheries. Smaller numbers were stocked by the Michigan Department of Natural Resources and Ontario Ministry of Natural Resources (OMNR). Until 1985, most Lake Trout stocked in Michigan waters were stocked along shore, and an average of 34% was distributed by boat. Offshore stocking became a more routine practice, and 85% were distributed by boat after 1989.

Marquette-Lake Superior was the principal strain stocked prior to 1990. After 1990, a wider variety of Lake Trout strains was used. Most Lake Trout were stocked as yearlings, although some fall fingerling Lake Trout were stocked. For reporting purposes, fall fingerling stockings were converted to yearling-equivalents by multiplying fall fingerling numbers by the fall-fingerling-to-yearling survival rate of 0.40 as in Elrod et al. (1988). All fall fingerling- and yearling-sized Lake Trout were given year-class-specific fin clips or lot-specific coded-wire tags. In 1985, refuges were designated at Drummond Island and 6-Fathom Bank (Figure 1) and these refuges became priorities for stocking. Lake trout of specific strains were requested for the refuges and marked with coded-wire tags (Ebener 1998; Johnson et al. 2004).

Stock Assessment Modeling

Since 1975, assessment of Lake Trout stocks has been done annually in Michigan's waters of Lake Huron with graded-mesh, multifilament, nylon gillnets set from late April through early June using gear and methods as described by Johnson et al. (2004). Survey stations were distributed in Michigan waters of three lake management units of the main basin of Lake Huron (Figure 1): north (Unit 1); central (Unit 2); and south (Unit 3). Data recorded from the Lake Trout catch included total length (nearest mm), weight (nearest 10 g), sex, maturity, occurrence of Sea Lamprey wounds, and stomach contents. Sea Lamprey wounds were classified according to King (1980). Most Lake Trout were assigned ages based on fin clips or coded-wire tags. Unmarked Lake Trout or Lake Trout with unrecognizable fin clips were aged using

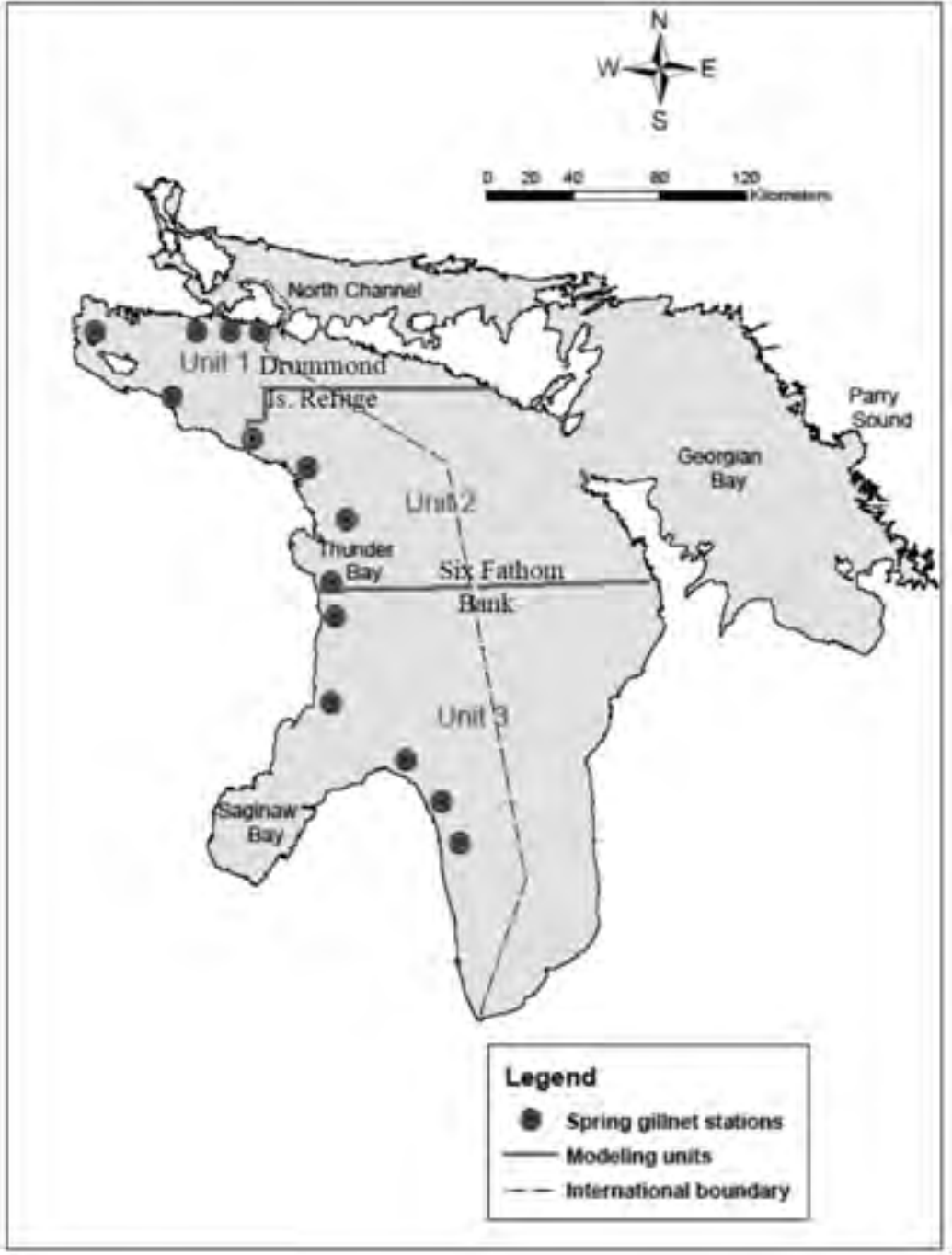


Figure 1. Lake Huron indicating modeling units and Lake Trout spring assessment sites.

scales, otoliths, or maxillae (Wellenkamp et al. 2015). Similarly, Lake Trout assessment data were collected and provided by OMNR (He et al. 2015).

Recreational harvest was monitored from 1986 to 2016 in Michigan waters by sampling completed-trip catches and recording fishing effort, using a stratified, randomized sampling plan as in Rakoczy (1997).

Recreational harvest of Lake Trout was insignificant in Ontario waters of Lake Huron’s main basin (Lloyd Mohr, OMNR, personal communication, 2013). Commercial harvest of Lake Trout by Michigan-licensed commercial fisheries was prohibited, but tribally-licensed commercial fishers in Michigan were permitted to harvest Lake Trout, principally in

Unit 1. Lake Trout were also commercially harvested by Ontario commercial fisheries in all three units. Commercial harvest was estimated based upon reported sales of Lake Trout. Age-specific biological data were taken from samples of the recreational catch and by dockside and on-board monitoring of the commercial catch. On-board monitoring was also used to correct for errors in reporting and for estimating Lake Trout discards and discard-induced mortality in the commercial catch (Johnson et al. 2004).

A portion of each Lake Trout year class was marked with coded-wire tags which were used to identify stocking site, year class, and strain. Coded-wire tags were sampled from the assessment, commercial, and recreational catch.

Statistical catch-at-age (SCAA) models were used to estimate and partition mortality rates and estimate abundance at age. Model design and technical details of the Lake Trout models were documented in Sitar et al. (1999) and Bence (2002). Initially, Lake Trout models were assembled for each of the three Lake Huron lake units, as reported by Johnson et al. (2015). Evidence of extensive movement between units 1 and 2 suggests these two units would best be treated as a single modeling unit. Thus, we show here results based upon combined modeling of units 1 and 2. Each SCAA was composed of three parts: (1) population model—population dynamics were driven by abundance at age in the first modeling year, annual recruitment, and annual, mortality rates, (2) predicted fishery catches, and (3) predicted catches at age. Fishery catches were modeled using Baranov's catch equation, while assessment catch per effort (CPE) and its age composition were modeled as proportional to population abundance and age structure. The above three parts of SCAA models were linked using predicted fishing and assessment effort, and estimates of catchability and age-specific selectivity in each gear. Most model parameters were adjustable, and best agreements between model estimates and observations were achieved by using a maximum likelihood approach to optimizing these variable parameters. Observed data included numbers and life stage of Lake Trout stocked, fishery and survey effort, fishery catches, survey CPE, age composition in fishery catches, age composition and contribution of wild Lake Trout in assessment catches, and some prior estimates of mortality rates.

Total mortality was compartmentalized into natural, fishing, and Sea Lamprey-induced. In older versions of the models, natural mortality estimates were allowed to vary by age group and over time. The most variation in mortality was among age-1 through age-3 fish. More recently, recruitment was estimated at age 3 and natural mortality was assumed to be constant for all recruited age groups. Fishing mortality was partitioned into commercial and recreational. Sea Lamprey-induced mortality was estimated separately, using wounding data from spring assessments (Sitar et al. 1999), and treated as input to the Lake Trout SCAA model. Recruitment of wild Lake Trout each year was estimated by fitting wild ratios at ages over years. A full description of the Lake Trout model is in Sitar et al. (1999) and Bence et al. (2002) with additional details in He and Cottrill (2012) and He et al. (2012a). Model output is expressed as maximum-likelihood estimates of these stock parameters. One strength of these models is explicitly recognizing uncertainties involved in stock assessment estimates. The uncertainties surrounding parameter estimates usually narrow with succeeding years of trend assessment. These models have been used every year in Lake Huron fisheries management since 2000 and models have been used in synthesizing food-web status in the main basin of Lake Huron (He et al. 2015). Our interpretations of modelling results were based on the consistency of direction in trends of population abundance and mortalities, rather than simply the parameter estimates for the most recent years. For simplicity, partitioned Lake Trout mortality rates are given here for age-7 Lake Trout only, the age at which they appear to be most vulnerable to commercial fishing. Including all year classes would be unwieldy but would likely have produced somewhat lower commercial fishing mortality rates than we give here.

Results

Stocking

From 1981 to 2016 a total of 55.8 million yearling-equivalent-sized Lake Trout was stocked in the main basin of Lake Huron, averaging 1,549,500 per year. Stocking rates increased somewhat after 1990 but otherwise varied little around this average (Figure 2). No refuge-area stocking occurred until

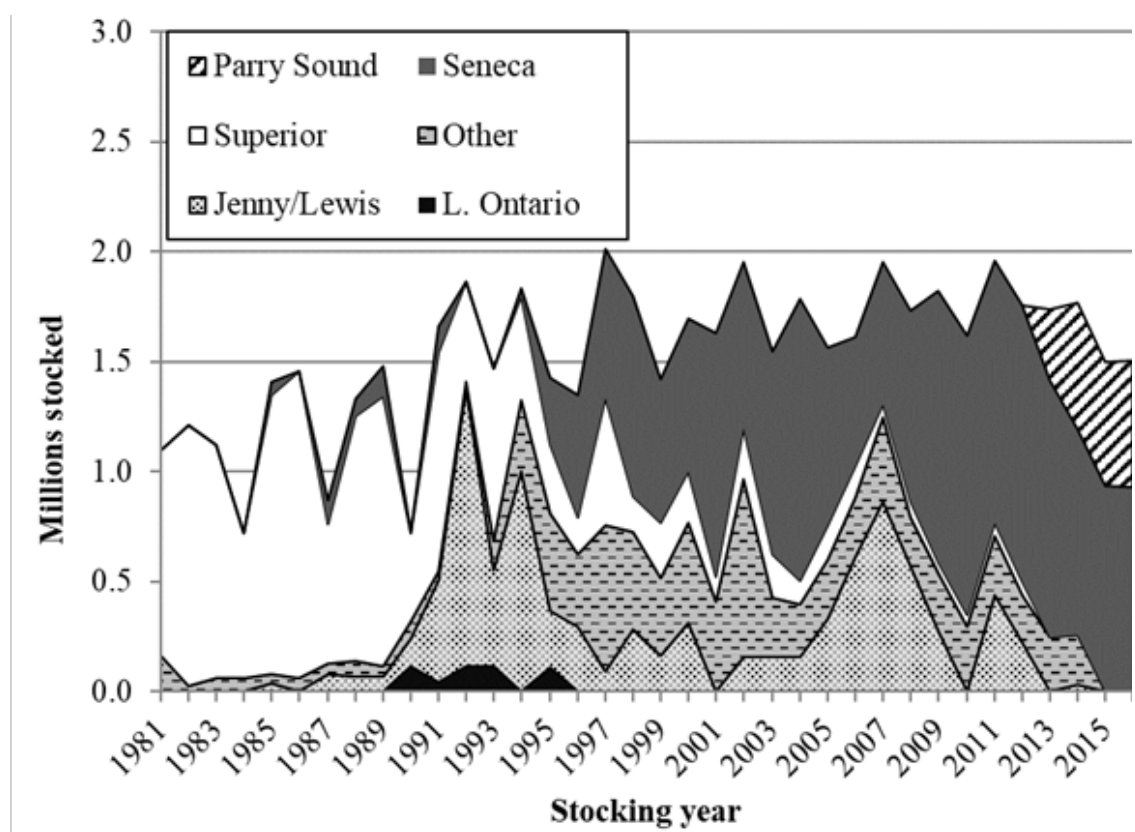


Figure 2. Lake trout stocking history,1981-2016, Main Basin of Lake Huron, Ontario and Michigan waters, by strain, expressed as yearling equivalents.

1986; from 1986 to 2001, refuges accounted for 24% of all stocking; stocking on refuges declined to 11% thereafter. Initially, strains of Lake Trout from Lake Superior-origin brood stocks composed almost the entirety of annual stockings. Beginning in 1985, Seneca strain, from Seneca Lake in the Finger Lakes area of New York, increasingly contributed. In the early 1990s, some Lake Trout of Lake Ontario origin were stocked; these were believed to represent a mixture of fish of Great Lakes, Seneca Lake, and other origins (Marsden et al. 1993). The contributions of Seneca strain steadily rose, especially in the north, composing 46% of all Lake Trout stocked in the main basin from 1995 to 2007 and 63% during the period 2008-2016. Strains originating from the Mackinaw area of lakes Michigan and Huron, which were stocked in Teton-Yellowstone National Parks during the late 19th century, were obtained from Jenny and Lewis lakes and stocked in the main basin beginning in 1985 (Figure 2). Beginning in 2013, the Lewis-

Jenny strains were replaced by a strain representing the remnant Parry Sound, Lake Huron, strain. Thus, the Lake Superior strains, which composed over 90% of the early stocking effort, were completely replaced by other strains by 2006. Ontario hatcheries used a different variety of strains than the U.S. and all Ontario strains are represented as “other” in Figure 2. Ontario ceased stocking the main basin after 2014. Over the time series, U.S jurisdictional stockings accounted for 85.6% of the numbers stocked; Canadian stocking accounted for 14.4%.

Lake Trout Status

From 1986 until 2000, total instantaneous mortality rates exceeded the target levels set for all units. Fishing was the leading source of mortality in combined Units 1 and 2 early during the time series, while Sea Lamprey and fishing were approximately equally important sources in all units from 1990 to 1999 (Figure 3). Consequently, estimated biomass of

Lake Trout age 7 or older (approximately spawning age) generally declined from 1984 to 1996 in the main basin of Lake Huron (figures 4 and 5); stocking rates were relatively stable during this period (Figure 2) while stocking success, defined as survival to age 7 per recruit, varied without trend (Figure 6). Natural reproduction was minimal during this period (Figure 5; Johnson et al. 2004). From 1997 through 2004, spawning-stock biomass steadily rose and the rise

was coincidental in time with declining Sea Lamprey wounding rates (figures 4 and 5). Concurrently, stocking success declined (Figure 6; He et al. 2012b) and contribution from natural recruitment began to rise (Figure 5). Biomass of age-7-and-older fish in combined units 1 and 2 peaked in 2013 and now appears to be stabilizing near 1.6 million kg, while the proportion composed of age-10-and-older fish and proportion of spawning stock from natural

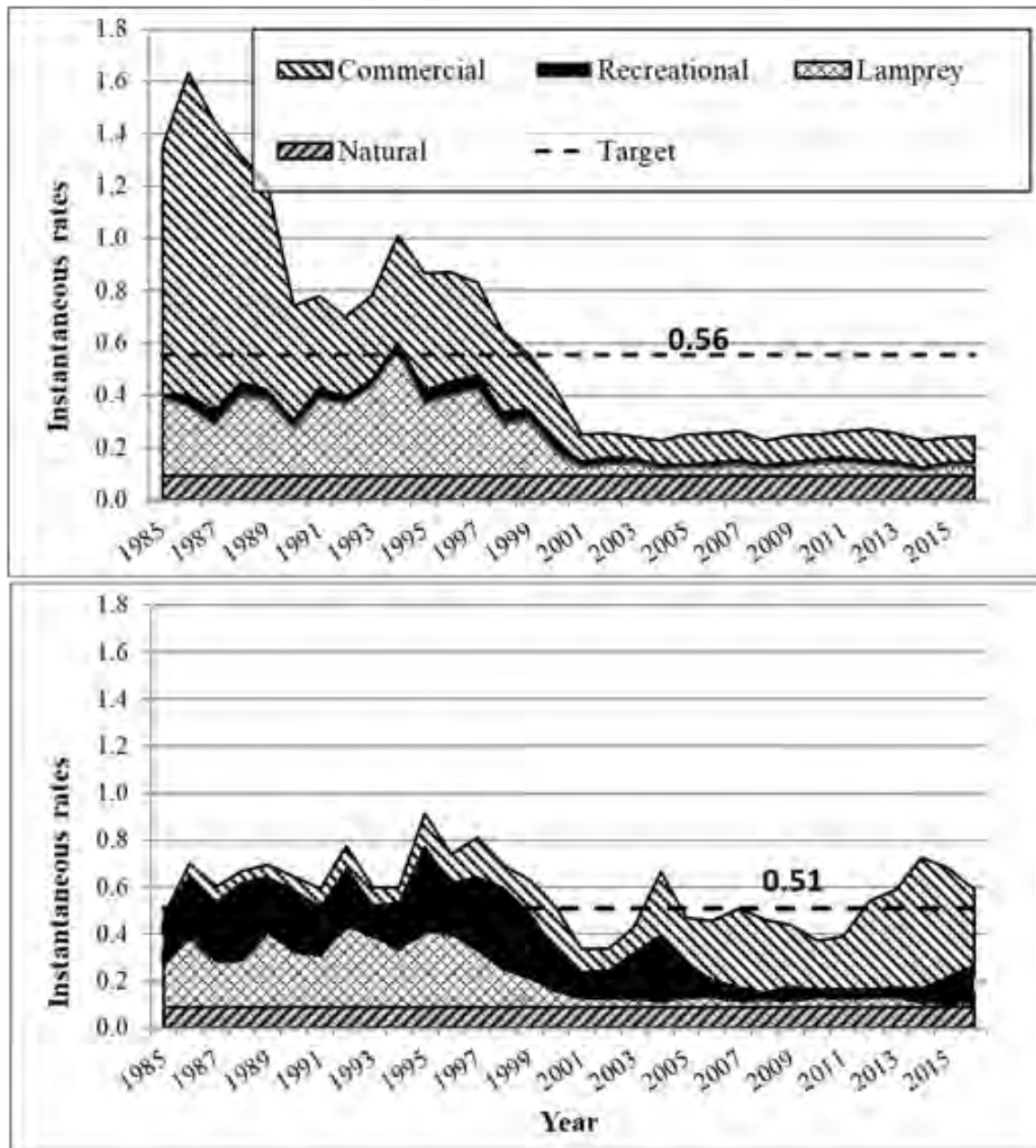


Figure 3. Sources of mortality at age-7 in the Main Basin of Lake Huron, Units 1 and 2 combined in upper panel; unit 3 in lower panel (Figure 1). Dashed horizontal lines represent total instantaneous mortality rates targeted by resource agencies.

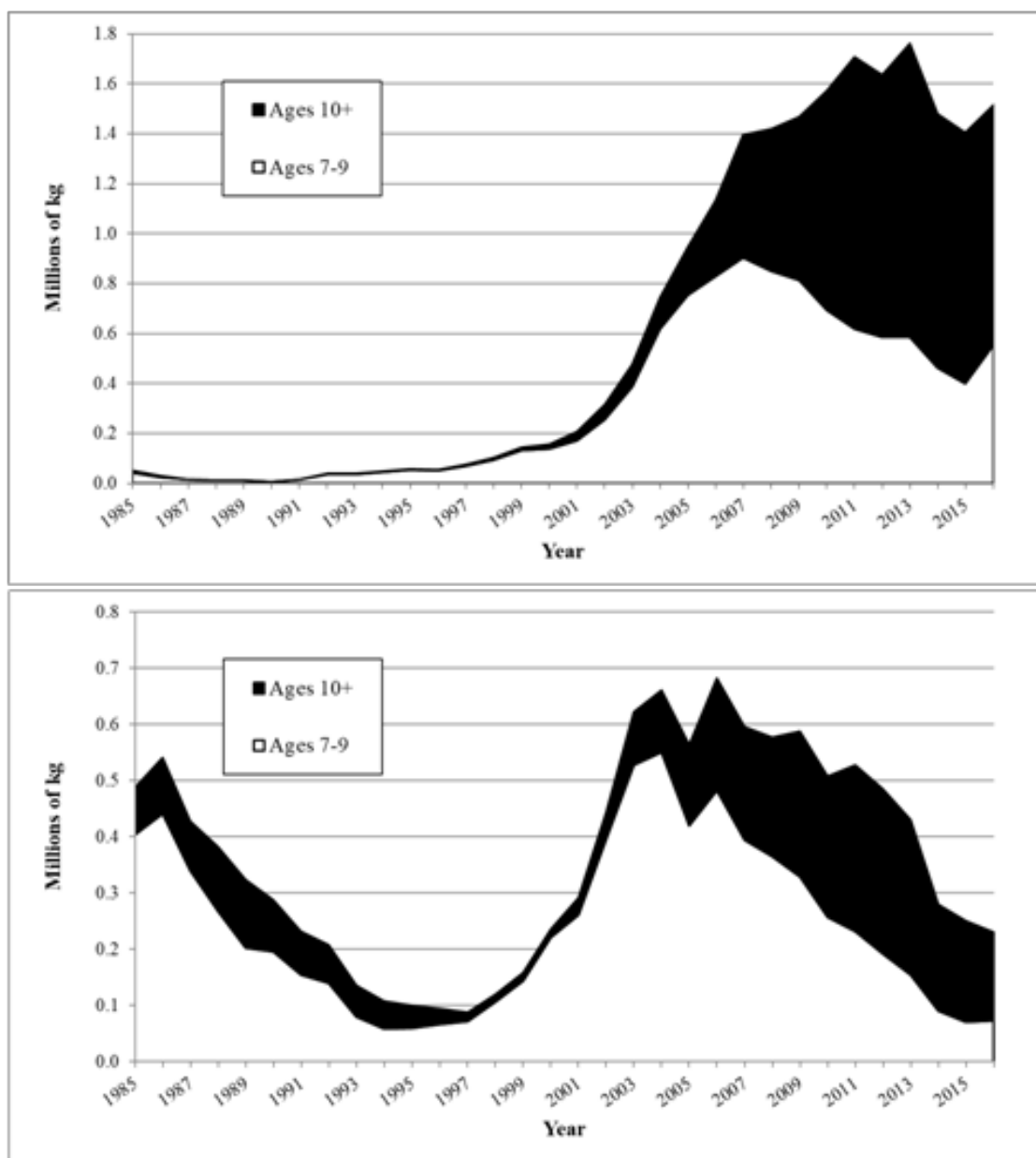


Figure 4. Trends in age 7-9 (including some immature fish) and age 10+ lake trout biomass in the Main Basin of Lake Huron. Units 1 and 2 combined in upper panel, Unit 3 in lower panel.

reproduction continued to rise (figures 4 and 5). Age-7-and-older biomass in the southern unit peaked at 0.68 million kg in 2006 and declined steadily thereafter. Biomass of age-10-and-older fish peaked in 2011 and declined steadily thereafter (Figure 4). The biomass of wild fish in the southern unit spawning stock peaked in 2006 but the proportion of wild-to-hatchery-origin fish there continued to rise due to a

steady decline in stocking success (figures 5 and 6). Stocking success in both units declined sharply after the 2006 year class and has ranged near-zero for the most recent year classes indexed (Figure 7). By 2011-2013, over 50% of main basin Lake Trout age-7 or younger were of natural, rather than hatchery, origin (Johnson et al. 2015); this proportion rose above 60% for the period 2011-2016 (Figure 7).

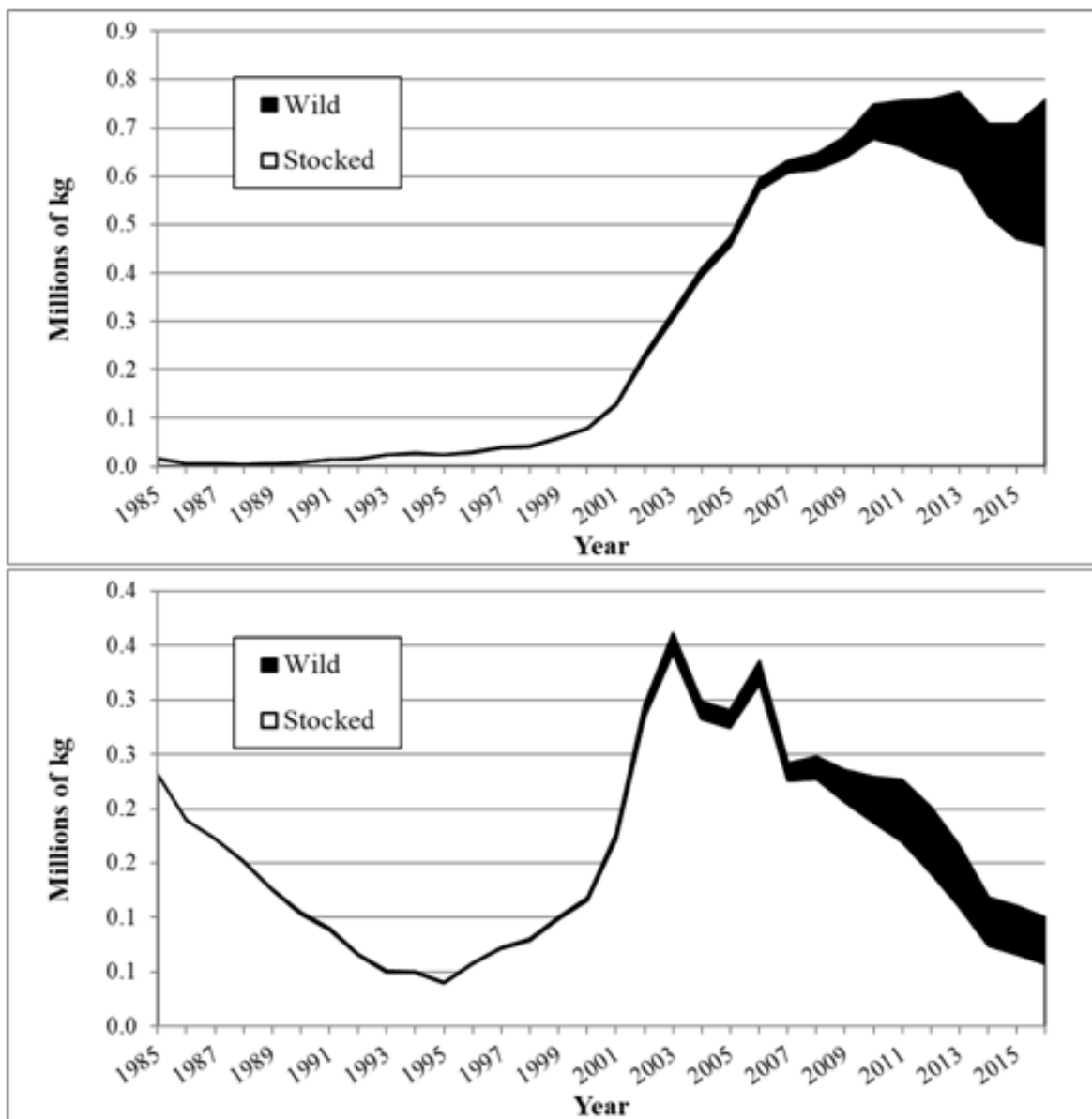


Figure 5. Trends in spawning-stock biomass of Lake Trout, Main Basin of Lake Huron, 1985-2016, (biomass estimates are lower than in Figure 5 because only mature fish are represented here). Units 1 and 2 combined in upper panel, Unit 3 in lower panel.

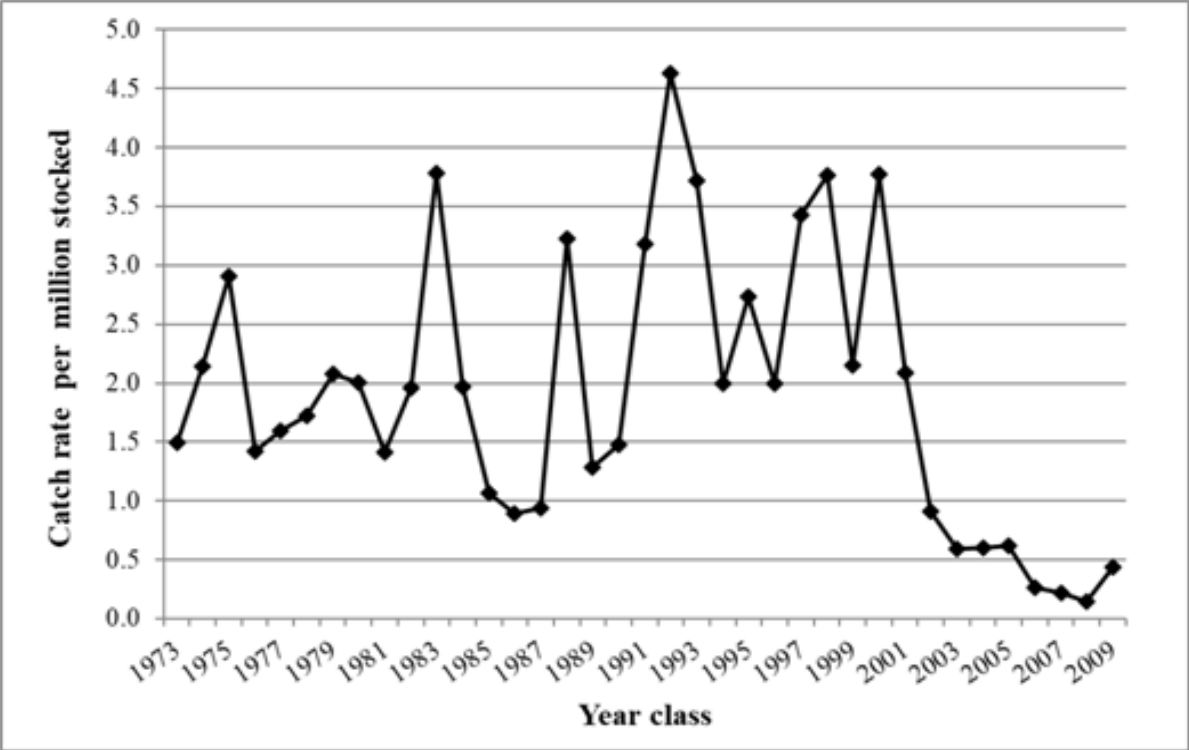


Figure 6. Geometric mean of number of age-7 lake trout caught per 1,000 m standard gillnet per million stocked (updated from He et al. 2012b).

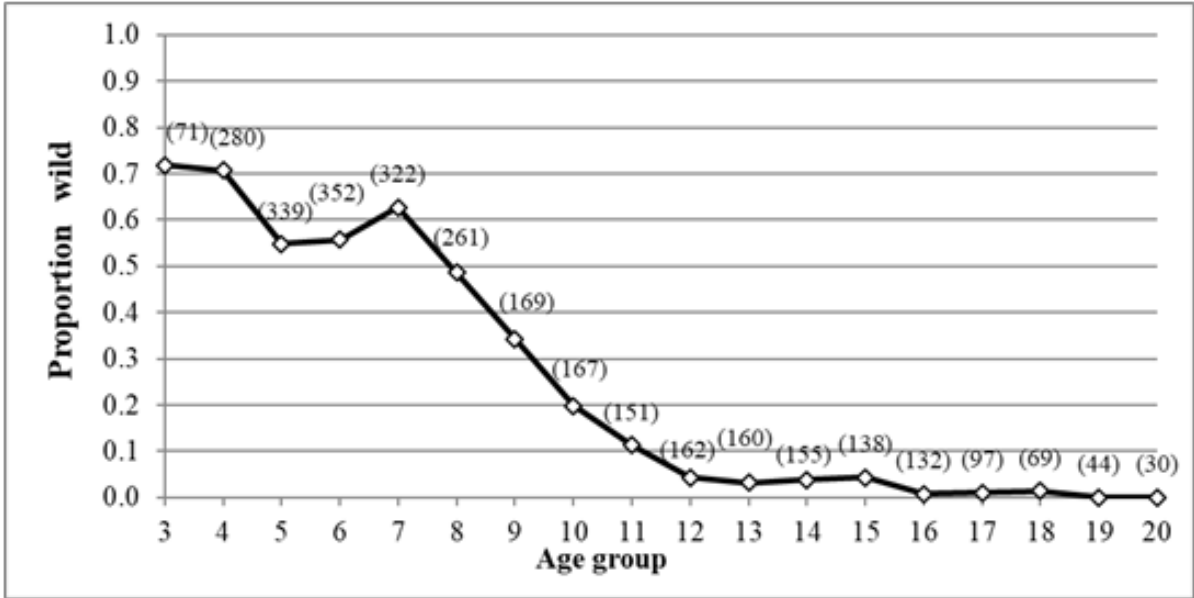


Figure 7. Proportion of Lake Trout of wild origin among fish sampled during 2011-2016 spring assessments, all units of the Main Basin of Lake Huron (sample sizes in parentheses).

Discussion

Johnson et al. (2004) concluded in 2003 that rehabilitation stocking had succeeded in building spawning-stock biomass of Lake Trout in Lake Huron, but there was little evidence of reproduction. By 2013, Lake Trout reproduction was evident lake wide and wild Lake Trout numerically composed approximately half the main basin Lake Trout population; more than half of Lake Trout younger than 7 years old were wild; prospects for rehabilitation appeared promising (Johnson et al. 2015). Here we show that all our metrics continue to show progress in the northern units (combined units 1 and 2) though not in the south. In the northern units mortality targets are being achieved, spawning-stock biomass remains robust, and recent recruits are dominated by wild fish. Stocking is no longer a significant population driver to any of the units. These developments represent a dramatic change since 2003. A number of impediments to reproduction were noted by Eshenroder et al. (1995), Johnson and VanAmberg (1995), Johnson et al. (2004), and Dobiesz et al. (2005). Actions that addressed these impediments appear to have enabled progress toward rehabilitation, particularly in the north. A discussion of these limitations to reproduction and how they may have been alleviated follows.

Two broad categories of impediments to reproduction were (1) inadequate survival to spawning age and, thus, low spawning-stock biomass, and (2) low viability or poor early survival of progeny.

Survival To Spawning Age

Excessive Sea Lamprey-induced and commercial-fishing-induced mortality caused the collapse of Lake Trout in the mid-20th century (Eshenroder et al. 1992) and inhibited the accumulation of Lake Trout spawning stock during the early rehabilitation period (Eshenroder et al. 1995; Johnson et al. 2004; Dobiesz et al. 2005, Johnson et al. 2015). By 1972, State-licensed commercial fishing for Lake Trout was prohibited in Michigan waters of Lake Huron (Tody and Tanner 1966; Rybicki and Schneeberger 1990). Commercial fishing for Lake Trout is, however, permitted in Ontario waters of the main basin. U.S. tribal commercial harvest of Lake Trout is permitted in northern Lake Huron, primarily in Unit 1, where a Consent Decree brought Lake Trout harvest under more effective quota management in 2000. Commercial fishing-induced mortality of Lake Trout

in Unit 1 declined incrementally from 1988 to 2000 (Johnson et al. 2004). Sea-lamprey induced mortality also declined after 1999 and, consequently, total mortality targets were achieved for the combined units 1 and 2 shortly after. Likewise, target mortality levels were achieved from 2000 to 2011 but rose above target thereafter in Unit 3. No commercial or recreational harvest of Lake Trout is permitted in either Drummond Island or Six-Fathom Bank refuges. Regulated recreational fishing is permitted in all waters of the main basin except the two refuges. A decline in recreational fishing effort in Michigan waters after 2004 (Johnson and Gonder 2010) drove a recent decline in recreational-fishing-induced mortality. Mortality rate has exceeded target levels in Unit 3 since 2011, however, principally driven by decline in recruitment and abundance. Commercial fishing has been the leading source of mortality in this unit since 2003 and both commercial- and recreational-fishing-induced mortality have increased in recent years. Yet, fishing effort and harvest have declined over this period of rising mortality rate (Michigan DNR, Alpena Fishery Station, unpublished data). The rise in fishing mortality rate was driven by declining recruitment and abundance in the southern unit, which drove fishing rates upward. Biomass of Lake Trout age 7-9 peaked in 2004 and has generally declined since. Biomass of Lake Trout age 10 and older peaked in 2012. The decline in biomass in Unit 3, driven by lower recruitment in the south than in northern Lake Huron, thus increased the vulnerability of this stock to prevailing fishing rates. Thus, while trends in total mortality have been generally positive, the decline in recruitment and consequent rise in fishing-induced mortality rate in Unit 3 could signal emerging issues there.

In addition to fishery closures and regulations, Michigan progressively expanded rehabilitation zones targeted for Lake Trout recovery by replacing a complex system of “primary”, “secondary”, and “deferred” zones (Eshenroder et al. 1995) with the three modeling units shown in Figure 1. Much of Lake Huron's historically most important spawning habitat in Unit 1 had been deferred from rehabilitation because of excessive fishing- and Sea Lamprey-induced mortality (Eshenroder et al. 1995). By 2000 these mortality sources had been alleviated (United States District Court 2000) and all Michigan waters of the main basin were targeted for rehabilitation,

thus affording corridors for movement of Lake Trout between habitat types; for example, between spawning habitats and feeding areas. As Lake Trout in Unit 1 recovered, modeling results suggested migration between units 1 and 2 was sufficiently high that the two units were best treated as a single model. The two modelling units were combined beginning in 2015. The better performance of models using larger modeling units and protection of all Michigan waters of Lake Huron for Lake Trout rehabilitation are consistent with the rationale of Foreman (2004) and Soulé and Noss (1998) for protecting corridors for restoration of wide-ranging keystone predators. The management of all Michigan waters of the main basin for Lake Trout rehabilitation accommodates the wide-ranging nature of this species (Adlerstein et al. 2007, Binder et al. 2017).

Madenjian et al. (2008a) and He et al. (2012b) concluded that more effective control strategies reduced Sea Lamprey-induced mortality rates of Lake Trout in the Drummond Island Refuge, but the beneficial effects were most apparent with the Seneca strain of Lake Trout, and less so with Lake Superior strains (Madenjian et al. 2008a). At Six-Fathom Bank Refuge, a mid-lake reef, Seneca strain Lake Trout composed the majority of spawning stock, while Lake Superior and Lewis Lake strains, which had also been stocked there, rarely survived to spawning age (Madenjian et al. 2004). Thus, while improvements in Sea Lamprey control contributed to the rise in spawning-stock biomass after 1998, the survival of Seneca strain was substantially more favored than that of other strains. Madenjian et al. (2008a) noted that Seneca strain fish appeared to be less vulnerable to commercial fishing gear, which further contributed to their greater survival as compared to Lake Superior strain Lake Trout.

Thus, the decision to employ Seneca Lake strain Lake Trout in the Lake Huron recovery effort appears to have been of crucial importance to the growth of Lake Trout spawning-stock biomass in Lake Huron's main basin. The Seneca strain was derived from gametes taken from Seneca Lake, New York, where Lake Trout are self-sustaining and Sea Lampreys have been abundant since at least the late 1800s (Royce 1949). The Seneca strain has also performed better than other strains in Lake Ontario (Schneider et al. 1996). Whether generations of exposure to Sea Lampreys conferred attributes that enhanced survival

or this variety of Lake Trout was innately more resistant to Lamprey effects is not clear. What is clear is that the Seneca strain survived and recruited to the spawning stock, in the presence of Sea Lampreys, in Lake Huron better than the other strains and was therefore vital to the progress toward recovery of Lake Trout in Lake Huron.

The combination of treatment of the St. Marys River Sea lamprey population (Schleen et al. 2003), progressively greater protection from fishing (Johnson et al. 2004; Dobiesz 2005), and stocking of Seneca strain in place of Upper Great Lakes strains of Lake Trout led to much better annual survival rates of older Lake Trout. Target mortality rates were achieved in all units by 2000 or shortly thereafter: spawning stock biomass peaked soon after 2000 and, except for Unit 3, has remained relatively high since.

Viability of Progeny

Evidence of reproduction of Lake Trout was synchronous in time across management units and approximated the timing of Alewife collapse (Riley 2007). Adult Alewives frequently consume larval fish (Kohler and Ney 1980; Wells 1980; Brandt et al. 1987), including fry of Lake Trout (Kruger et al. 1995; Madenjian et al. 2008b). Furthermore, Alewives contain thiaminase, a thiamine-degrading enzyme. Diets rich in Alewives can lead to egg thiamine deficiency that in turn has been associated with mortality of larval Lake Trout and other salmonid species (McDonald et al. 1998).

A combination of Alewife-mediated thiamine deficiency and predation by Alewives on fry (as theorized by Madenjian et al. 2008b) probably contributed to the suppression of Lake Trout reproduction in the main basin of Lake Huron. Until about 2004, Alewives contributed importantly to diets of Lake Trout in the main basin of Lake Huron (Dobiesz 2003; Diana 2007) and composed 75% to 90% of the diets of Lake Trout on the Six-Fathom Bank Refuge and Yankee Reef, an offshore reef near Six-Fathom Bank (Madenjian et al. 2006a). Egg thiamine levels of Lake Huron main basin Lake Trout were low enough in 2001-2004 to have caused lethal and sublethal effects on progeny of at least 50% of Lake Trout spawning stock, but egg thiamine rose after 2004. Egg thiamine levels after 2007 were high enough at main basin stations to no longer pose a serious impediment to Lake Trout reproduction

(Riley et al. 2011). Thus, the collapse of Alewives appears to be a contributing factor in the rising rate of reproduction across the main basin of Lake Huron.

In summary, it is likely that a combination of heightened protection from harvest, more effective control of Sea Lampreys, and the collapse of Alewives was responsible for the progress since the late 1990s toward rehabilitation of Lake Trout in Michigan waters of Lake Huron. Stocking could not meet its objectives until these issues were addressed. Beginning in 1995, Seneca strain Lake Trout became one of the principal strains employed in stocking; the evidently lower vulnerability of Seneca strain Lake Trout to Sea Lampreys and gillnet fisheries was another important contributing factor to recovery.

By 2016 over 60% of the Lake Trout less than age 8 in the main basin were wild. Stocking is now contributing little to recruitment. Consequently, fishery managers are planning reductions in stocking to begin in 2018.

He et al. (2015) measured rising predation rates in Lake Huron since Alewife collapse. By 2007 Lake Trout and Walleyes *Sander vitreus* had replaced Chinook Salmon *Oncorhynchus tshawytscha* the lake's leading predators. As the prey base declined, consumption continued at the same pace as during the pre-Alewife-collapse era. Thus, by approximately 2007, the stocking and management of native piscivores not only reestablished spawning populations large enough to foster natural reproduction but also restored ecosystem functions of top predators. Grooming, or 'cultivation', as in Walters and Kitchell (2001), of the prey base by these top predators has thus far prevented a recovery of the Alewife population (He et al. 2015). We also point out that the reduction in Alewife growth and condition due to reduced Diporeia abundance (once a leading prey item for Alewife) associated with the dreissenid mussel invasions of Lake Huron served to exacerbate the predation effect on Alewives by the top predators (Madenjian 2006b; Pothoven and Madenjian 2008). Consequently, the present-day fish community composition sans Alewife in Lake Huron appears to favor reproduction of Lake Trout.

Management Implications

Rehabilitation of Lake Trout now seems nearly accomplished, particularly in the north. Consequently,

managers have prescribed sharp reductions in stocking. However, sustainability of this recovery is not ensured. Reductions in funding for Sea Lamprey control or adaptive responses by Sea Lampreys to control measures could compromise the effectiveness of this vital management tool. Recent declines in availability of Lake Whitefish, the primary species targeted by upper Great Lakes commercial fisheries, could lead to greater targeting and incidental catch of Lake Trout. Total mortality targets for Lake Trout were only recently achieved. Ominously, fishing mortality rate rebounded and has exceeded target levels in the southern unit (Unit 3) since 2011. The foundation of the progress we report here has been effective management of mortality rates, which in turn is predicated upon sustained, effective Sea Lamprey control and management of commercial and recreational harvest.

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Cutthroat Trout in Saltwater: Spawn Timing, Migration Patterns and Abundance of Anadromous Coastal Cutthroat Trout

James P. Losee, Gabe Madel, Hannah Faulkner, Andrew Claiborne,
Todd R. Seamons, William Young

Washington Department of Fish and Wildlife, 600 Capitol Way N. Olympia Washington 98502

Abstract—Despite being one of the most widely distributed salmonids along the Pacific coast, the Coastal Cutthroat Trout *Oncorhynchus clarkii clarkii* is one of the least understood. In 2007 we began a multidisciplinary project to clarify the spawn timing, spawner abundance, redd morphology, marine migration, and genetic population structure of anadromous Coastal Cutthroat Trout in Puget Sound. Using PIT (Passive Integrated Transponder) tags, genetic stock assignment and scale analysis combined with redd surveys, we have documented important insights into the biology of anadromous Cutthroat Trout. The majority of “sea-run” Cutthroat Trout enter marine waters at age 2, exhibit high site fidelity to nearshore beaches as juveniles and adults and return to natal tributaries in the spring to spawn (February through June). Migration distances are limited with observations of high site fidelity year-round for juveniles and adults but interestuarine movements were common. Together, this new information provides fisheries managers with improved tools to maintain healthy populations of anadromous Cutthroat Trout across their range.

Introduction

The Cutthroat Trout *Oncorhynchus clarkii clarkii* has been described as the ancestral salmonid in the Pacific Northwest (Trotter 2008), and through thousands of years of probing inland and southward, this species has evolved into at least 11 other subspecies and more than five life history types, including anadromy (Behnke 1979). The subspecies Coastal Cutthroat Trout is not an important commercial species and so is understudied relative to other salmonids on the west coast of North America. Although general life cycle information has been documented for anadromous Coastal Cutthroat Trout (Wenburger 1998; Trotter 2008), their spawn timing, migration patterns and status are poorly understood. Without this information, biologists may be unable to evaluate management plans or ensure the long-term stability of a population.

In the absence of definitive information on the status of Coastal Cutthroat Trout, managers have relied on conservative management approaches to minimize fishing mortality in hopes of maintaining or increasing the number of Cutthroat Trout while continuing to offer fishing opportunity. While harvest is permitted in selected rivers in Washington State, current sport fishing regulations for Cutthroat Trout in marine waters require barbless hooks and prohibit

harvest year-round. Unlike harvest fisheries, where the majority of fish captured are removed from the population, catch-and-release fisheries assume that the majority of fish captured are successfully returned to the water alive and thus remain part of the population. Hooking mortality associated with catch-and-release fisheries is thought to be low (Schill et al. 1986), however, mortality rates may vary widely depending on a variety of factors (e.g., gear type, angler experience and environmental conditions; Gresswell and Harding 1997). Under a high mortality rate scenario and where stocks of concern are frequently encountered by anglers, fishing pressure could conflict with conservation concerns, regardless of special angling regulations such as catch and release.

In marine waters Cutthroat Trout are managed assuming a mixed-stock management type, but the degree of mixing and general migrations patterns are unknown. Cutthroat Trout exhibit high site fidelity during spawning (Wenburger and Bentzen 2001), forming genetic stock structure organized at the stream level. Results from tagging studies in Hood Canal, a large fjord of Puget Sound, suggests that Cutthroat Trout rarely migrate far from their natal stream in the marine environment (Moore et al. 2010). It is unknown, however, whether or not the fidelity Cutthroat Trout exhibit to their natal inlet in Hood Canal is characteristic of Cutthroat Trout throughout

Puget Sound. If so, the popular sport fishery concentrated near estuaries adjacent to Cutthroat Trout spawning streams may be best managed as a series of inlet-specific terminal fisheries where angling regulations can be applied to marine waters based on the status of the associated population. Conversely, if longer distance migrations that are common for other species of anadromous trout (i.e. Bull Trout *Salvelinus confluentus*, Brown Trout *Salmon trutta*, Arctic Char *Salvelinus alpinus* etc.; Quinn and Myers 2004) are observed for Cutthroat Trout, inlets of Puget Sound may be best characterized as mixed stock fisheries. In this case, angling regulations applied across a broad geographic region may be appropriate to protect small, independent populations mixed with larger ones.

The goal of this work was to characterize Cutthroat Trout found in the marine and freshwater environments of south Puget Sound and provide fisheries managers with tools to improve monitoring and management activities. Specifically, we sought to describe the (1) spawn timing and abundance, (2) size and age, and (3) migration patterns of anadromous Cutthroat Trout in South Puget Sound Washington.

Combined, this new information may assist managers in designing fisheries to address stocks of concern and promote the long term viability of anadromous Cutthroat Trout.

Methods

Study Area

Puget Sound, Washington, is characterized by numerous fjord-like inlets each fed by one or more streams draining into it. As a whole, Puget Sound has water chemistry properties resembling partially mixed estuaries (Sutherland et al. 2011). The current study was conducted in freshwater and nearshore marine waters of South Puget Sound that represent high-use fishing areas for those targeting Cutthroat Trout (Lothrop and Losee 2016). The marine study areas comprised Skookum, Totten and Eld inlets as well as the area where these three inlets meet (Squaxin Passage, Figure 1). The aspects of the study carried out in freshwater included the three major streams draining the marine study area, Skookum Creek, Kennedy Creek, and McLane Creek.

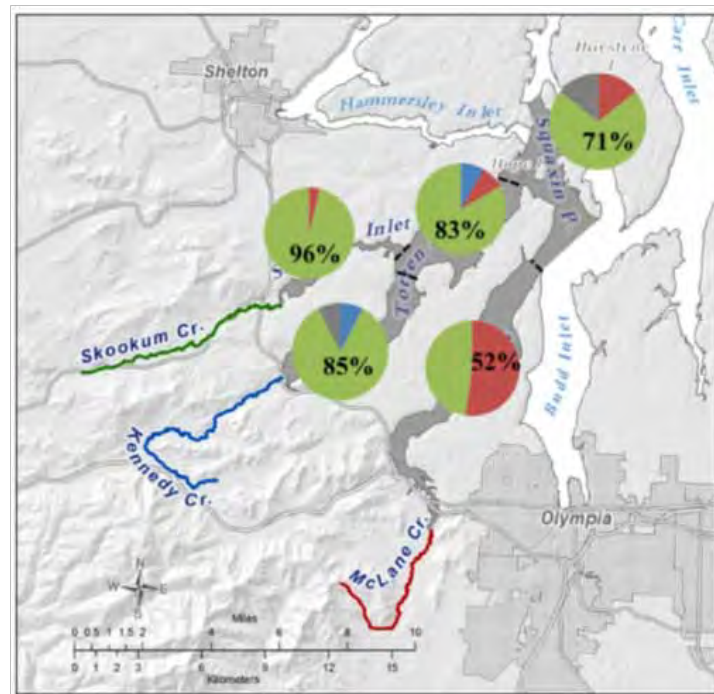


Figure 1. Study area in South Puget Sound, Washington, U.S.A and proportional contribution of genetic assignment of Coastal Cutthroat Trout by sampling region. Colored lines indicate streams included in baseline samples for genetic stock assignment. Colors within pie graphs indicate natal stream assignment; Skookum Creek: green, Kennedy Creek: blue, McLane Creek: red and unidentified source population: grey. Percentages indicate stock with greatest contribution by region.

Spawning and Abundance

To evaluate spawning timing of Coastal Cutthroat Trout the Skookum Creek index area (RKM 8.9 to 12.1) was surveyed from October to June 2008–2014 using standardized salmonid redd survey methodology described by Gallagher and Gallagher (2005). Each redd was flagged with the date, the surveyor's initials, and other descriptive details as needed. The same two trained individuals were assigned to survey redds for the life of the study, with few exceptions, thus reducing interobserver error and allowing for a comparison of relative abundance across various time scales (i.e., days, months, and years).

Measurements of redd size and sediment type were collected in 2014. Pit length was the total length of the pit as measured parallel to the streamflow. Pit width was the maximum width of the pit as measured perpendicular to the streamflow. The tail spill is the sediment that is excavated by spawning fish and elevated above the stream bed immediately downstream of the pit; its length was measured as the total length parallel to the streamflow. The diameters of substrate particles adjacent to each redd were measured with a metric rod to evaluate substrate composition.

To estimate escapement we sought to convert total redd counts to an estimate of mature Cutthroat Trout. To estimate the number of fish per redd, we tagged fish > 200 mm (FL) with PIT tags in the marine study area and intercepted a proportion of those fish in the Skookum Creek spawning index area during the spawning season using fixed PIT tag antennas. Fish detected on antennas in the index area represented a proportion of the total number of fish entering the index area. An estimate of the total number of fish entering Skookum Creek was produced by estimating the proportion of total Skookum Creek Cutthroat Trout tagged through monthly sampling of Cutthroat Trout in the Skookum Creek Estuary using a beach seine. We expanded the number of tagged Cutthroat Trout detected on fixed antennas by the estimate of the proportion of tagged fish from monthly marine sampling to achieve an estimate of the total number of Cutthroat Trout entering Skookum Creek. This number was then divided by the total number of redds, producing the estimated fish per redd. We then multiplied the number of fish per redd by the total number of redds in the Skookum Creek drainage to produce an escapement estimate for Skookum Creek.

Collection of Fish and Age Analysis

Fork lengths (FL), scales and tissue samples were collected from Cutthroat Trout throughout the study area using hook and line and beach seine. To determine age, scales collected from the preferred area above the lateral line midway between the dorsal and adipose fins were analyzed for age and saltwater entry at the Washington Department of Fish and Wildlife (WDFW) marine aging lab. The scales were mounted on scale cards, lightly dyed for visibility and analyzed under 40× magnification. We defined juveniles as those without an annulus on their scales and adults as those with no annulus or ≥ 1 annuli after marine entry.

Marine Movements

To describe both broad and fine scale movements, we used genetic methodologies as described by Losee et al. (2017). To document broad patterns of stock-specific movements and identify the degree of “mixing” of various populations, we sampled Cutthroat Trout throughout the study area using hook and line and assigned catch to their population of origin using genetic stock identification. To describe fine-scale movements of Cutthroat Trout, we evaluated site fidelity by sampling the same location in Eld Inlet monthly, using a beach seine while recording the number of times individual Cutthroat Trout were recaptured at this location. Recaptures were identified using genetic tags; samples with matching genotypes were assumed to be the same individual.

Results

Spawning and Abundance

During 2009–2014, we observed 544 Coastal Cutthroat Trout redds and 148 live Coastal Cutthroat Trout. Coastal Cutthroat Trout redds were observed in the index area as early as February 2 and as late as May 27 (Figure 2). The observed Coastal Cutthroat Trout spawning period ranged from a minimum of 47 d in 2009 to a maximum of 114 d in 2012 (mean $\pm$ SD = 79.9 $\pm$ 21.0 d). Mean pit length was 0.48 $\pm$ 0.14 m (mean $\pm$ SD), and mean pit width was 0.43 $\pm$ 0.14 m. Coastal Cutthroat Trout redds tended to be constructed in habitat that was dominated by small gravel (~69%;) but large gravel and small cobble were also common.

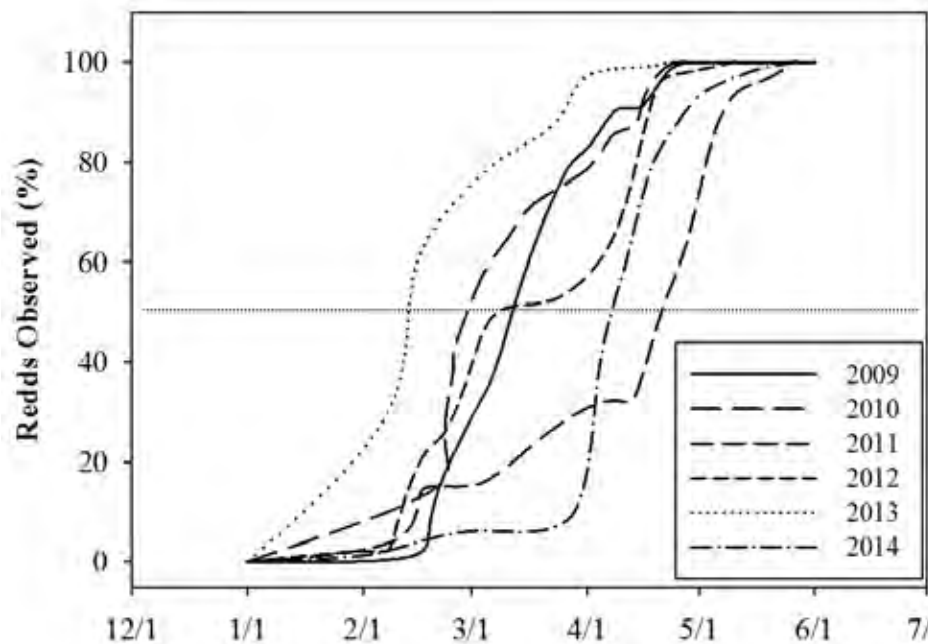


Figure 2. Cumulative percentage of anadromous Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*) redds observed in Skookum Creek by date. Data points from surveys are joined by a smoothed line. Horizontal dotted line identifies the date by which 50% of redds had been observed.

Using Passive Integrated Transponder (PIT) tags in the spring of 2017, we estimated that 91 adult Cutthroat Trout entered the index area of Skookum Creek. During the same time period, we enumerated 74 redds. With one year of data, the preliminary estimate of number of fish per redd is 1.23. Using this estimator and expanding redd counts to include spawning habitat in Skookum Creek outside the index area, we estimated an average escapement of anadromous Cutthroat Trout for Skookum Creek during the study period (2008-2015) of 132 (± 39.5 S. D.). The estimate of fish per redd will be further evaluated in 2018 and 2019 allowing for a robust estimate of fish per redd and total escapement of Cutthroat Trout for the entire study period.

Age Distribution

Based on scale analysis, mature Cutthroat Trout sampled in the marine environment were dominated by 3 year olds (35.0%, Figure 3). Nine separate life history strategies were identified with juveniles entering marine water at age 1, 2, and 3. Few fish demonstrated a spawning check ($N=4$) at the time of capture in marine

water and three of these Cutthroat Trout spawned 2 years after entering the marine water while one individual had spawned after only 1 year in saltwater.

Marine Movements

Inlets of South Puget Sound were comprised of multiple genetically distinct populations in all months. The majority (71.6%) of Cutthroat Trout captured using hook and line in the marine environment were less than 15 km from the mouth of their natal stream while 14.1% were captured greater than 30 km from their natal stream. Average migration distance was greatest in summer months when marine temperatures are greatest and spawning season has ended.

Following the initial sampling event in January, we identified genetic matches (recaptures) in every month of the study with the exception of the month of June when no Cutthroat Trout were captured (Figure 4). Overall, 21% of Cutthroat Trout sampled in this study were encountered during subsequent sampling events. Highest recapture rates occurred on March 26, 2015. On this sampling event, all adults captured had been sampled previously ($N=24$) and 86% of total

catch (juveniles + adults) had been captured previously (25/29). During the course of the study, 13.1% of known juvenile Cutthroat Trout captured at the study site were captured more than once and 30.8% of adult Cutthroat Trout were captured more than once.

Discussion

We documented that Coastal Cutthroat Trout spawning activity was protracted over an extended time period and exhibited a high degree of interannual variability relative to other salmonids. Numerous studies have used mark–recapture methods and/or trap counts to describe the timing of the Coastal Cutthroat Trout migration into freshwater for populations across the subspecies’ range. Although this information is valuable for estimating the time of adult freshwater entry and for describing habitat use, generating definitive information on the spawn timing of Coastal Cutthroat Trout is imperative for successful management. By comparing weekly counts of Coastal Cutthroat Trout redds to estimates of abundance using PIT tags, we were able to provide an accurate estimate of spawn timing and estimate the number of fish present during the construction of redds within the index area. A logical next step would be to replicate this work across other systems in Puget Sound and beyond. However, information reported here should serve as a starting point to allow managers and volunteers to estimate abundance of Cutthroat Trout across their range.

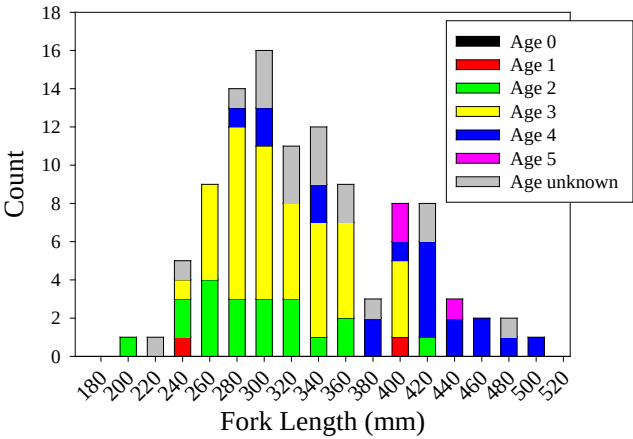


Figure 3. Length frequency distribution and age composition (stacked bars) of Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*) captured in marine water of South Puget Sound, Washington.

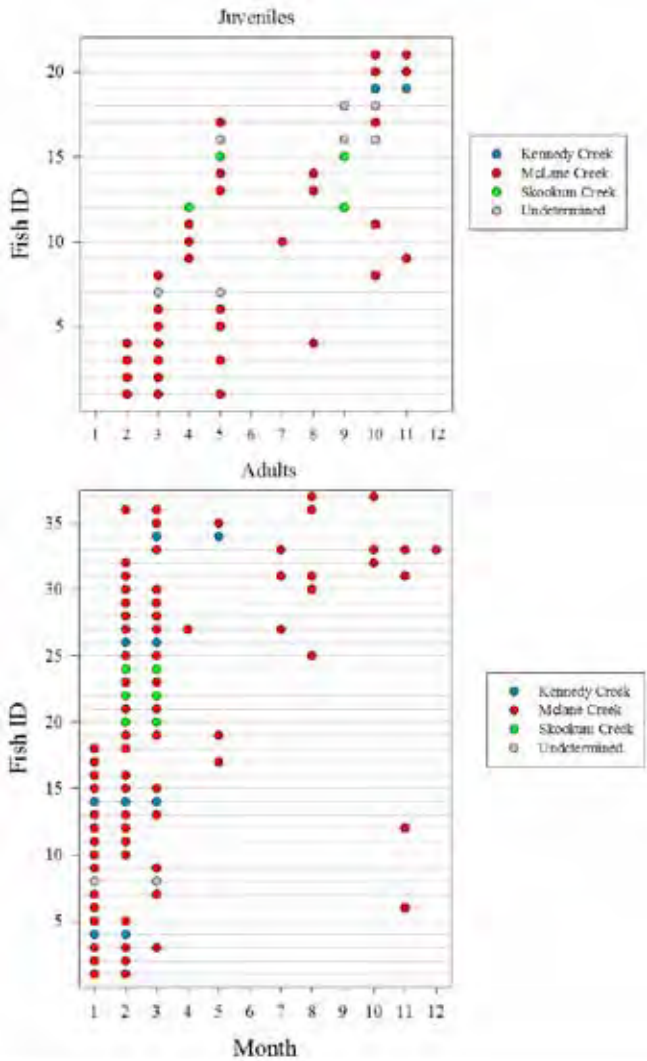


Figure 4. Month of recapture for Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*). Each horizontal line represents an individual Coastal Cutthroat Trout (Fish ID) captured more than 1 time in Eld Inlet, South Puget Sound Washington in 2015. Dots indicate months of capture (x axis). Colors indicate genetically assigned stream of origin.

With genetic stock identification, we showed that anadromous Coastal Cutthroat Trout regularly made marine migrations outside of natal inlets. Anadromous trout exhibit a variety of different migration patterns from transoceanic migrations of Steelhead (Quinn and Myers, 2004) to short interbasin migrations of Dolly Varden *Salvelinus malma* (Spares et al. 2015) and Brown Trout *Salmo trutta* (Eldøy et al. 2015) as well as partial expression of anadromy in Dolly Varden and Rainbow Trout (Bond et al. 2015). While studies specifically focused on Cutthroat Trout in

the marine environment are limited, the majority of what is known suggests that Cutthroat Trout make relatively short distance marine migrations (Goetz et al. 2013), do not migrate across large bodies of water (Jones and Seifert 1997) and are unlikely to leave their natal inlet (Moore et al. 2010); however, longer migrations have been documented, e.g., Pearcy et al. (1990). In the current study, the majority of fish were assigned to their nearby natal streams <15 km away from the capture location; however, a second mode of longer migrating fish was observed. Fish originating from Skookum and McLane creeks were recovered in high proportions on the margins of the study area, suggesting that it is likely that the full extent of the marine distribution of our study populations was not observed. These results are consistent with those of Goetz et al. (2013) where most fish underwent short marine migrations (residents) while others exhibited longer migrations (migrants). Overall, information reported here suggests that, unlike Cutthroat Trout observed in Hood Canal (Moore et al. 2010), Cutthroat Trout in South Puget Sound regularly leave their natal inlet and exhibit a high degree of variability in migration distance.

Along with interestuarine migrations, we found that Cutthroat Trout exhibited high site fidelity in an area where they are easily accessible to anglers. Recent work by WDFW has identified challenges in management of anadromous Cutthroat Trout due to their mixed stock composition in marine water (Losee et al. 2017), unpredictable migratory patterns (Moore et al. 2010), variability in spawn timing (Losee et al. 2015), and increasing effort by sport anglers targeting them. As a result of much of the uncertainty surrounding anadromous Cutthroat Trout, Washington State manages them conservatively, relying on catch-and-release regulations to minimize fishing mortality. Results of the current research clarify movement patterns of this species and add additional support for conservative regulations to protect Cutthroat Trout from overharvest in areas where remaining nearshore habitat overlaps with fishing access sites. Additionally, catch-and-release regulations most likely provide the greatest economic benefit by maximizing catch rates over the long term for relatively small population sizes.

It is now understood that sport fishers targeting Coastal Cutthroat Trout in marine waters of South Puget Sound encounter a variety of distinct stocks, each made up of less than 300 fish. It is also known

that large bodied (>350mm) individuals that had spawned previously were uncommon. Depending on the goals of fish managers, considerations for more fecund, larger females and those stocks that are limiting should be made when designing regulations, consistent with a mixed-stock management strategy. While this may not be feasible due to limited funds and federally mandated recovery efforts for higher priority, Endangered Species Act listed stocks, methodologies for estimating abundance of Coastal Cutthroat Trout in-river are now available and can be implemented with little cost. Finally, by gaining additional information on the movements, age structure, and life history of anadromous Cutthroat Trout relative to historical information, managers may be better able to evaluate the impact sport fisheries have on Coastal Cutthroat Trout in the marine and freshwater and design fisheries to maximize long-term fishing opportunity on abundant stocks.

Acknowledgements

We are thankful for the continued support of the Coastal Cutthroat Coalition and all its' contributors; specifically Greg Shimek, Mark Dalton and Richard Malzahn. We are also grateful for Riley Freeman for his involvement in this project since the beginning and the inspiration and scientific expertise from Larry Phillips to better understand this unique species.

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Using Research and Monitoring Data to Prioritize Native Trout Conservation Work in the Teton River

Mike Lien

Friends of the Teton River, P.O. Box 768, Driggs, Idaho 83422

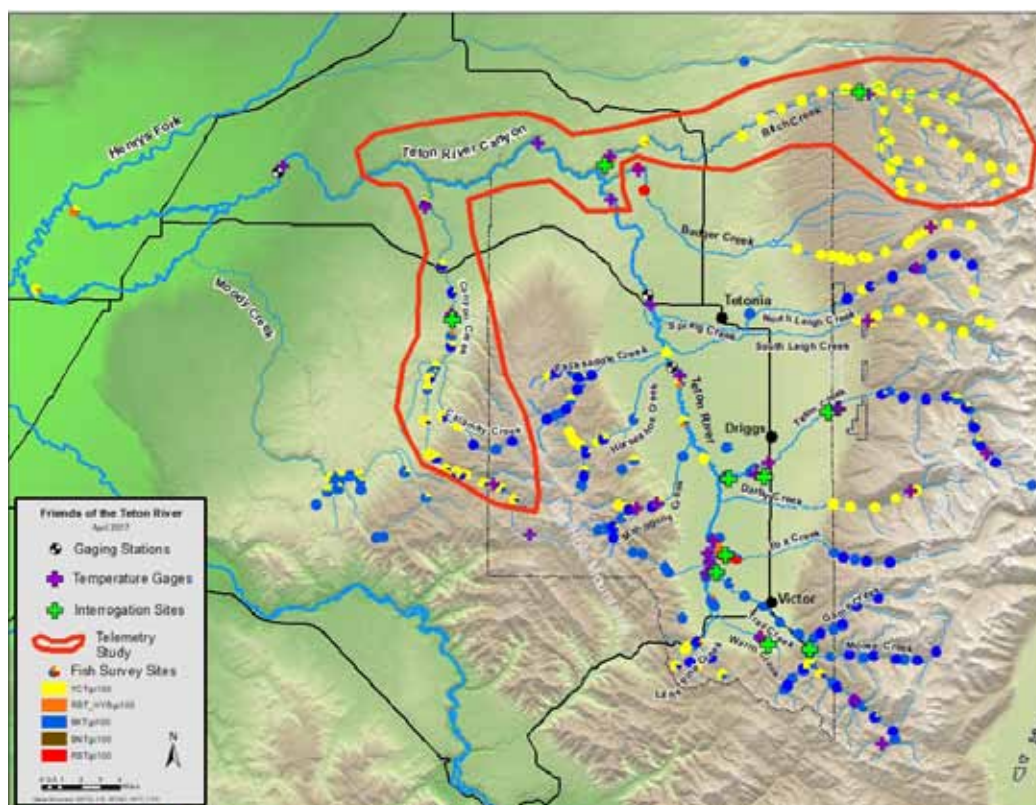
Extended Abstract

In 2017, Friends of the Teton River completed the Teton Watershed Fisheries Research Project which summarized 12 years of comprehensive fisheries, stream temperature, habitat, and hydrology monitoring data (2003-2015) (Figure 1). The most significant research findings included the following: (1) trout populations, including native Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT) have improved dramatically in the Teton River since 2003, mostly likely due to conservation work implemented by FTR, Teton Regional Land Trust, and landowners which started in earnest in the same year; (2) each significant YCT population has been identified; (3)

factors contributing to native trout success in certain areas of the watershed and the factors contributing to native trout declines in other areas have been determined; and (4) restoration strategies designed to specifically benefit YCT have been developed. These findings have been used to supplement the agency data base, prioritize native YCT conservation projects, and to evaluate the efficacy of YCT conservation projects in the watershed.

Conservation work in the watershed started in earnest in 2003 and includes fish passage projects, fish screens, stream restoration projects, flow restoration transactions, canal management improvements, and

Figure 1. Fisheries, temperature and flow monitoring sites in the Teton River watershed, Idaho.



riparian fencing projects. To measure the success of these projects on a watershed scale, FTR compared the cumulative total of the number of completed conservation projects across years to fish density data from Idaho Department of Fish and Game (IDFG) Teton River electrofishing sites (Figure 2). In the Upper Teton River, trout densities increased dramatically between 2003 and 2015 at the IDFG Nickerson and Breckenridge electrofishing sites (Figure 3). At the Nickerson Site, which is just downstream from the majority of conservation projects, trout densities increased nearly ten-fold from 450 trout/mi to 4,100 trout/mi. Combining the most frequently surveyed Upper Teton River IDFG sites (Breckenridge and Nickerson), YCT populations have also increased dramatically for the same time period from 10 YCT/mi to 366 YCT/mi (Figure 4).

In tributaries with only YCT and without nonnatives where conservation projects have occurred, YCT densities have increased, although the data set for these tributaries is limited (Figure 5).

Fisheries management impacts were measured by IDFG, and it was determined that management and angler impacts were not responsible for the increase in fish densities which led Dan Garren, the IDFG Regional Fisheries Manager, to conclude that conservation work was probably the reason for the increase in densities. Hydrology and stream temperature data were analyzed by Robert Al-Chokhachy (US Geological Survey) for the same time period and it was determined that these factors were not responsible for the increase in fish densities either; therefore, it was concluded that conservation projects were most likely the reason for the increase in densities.

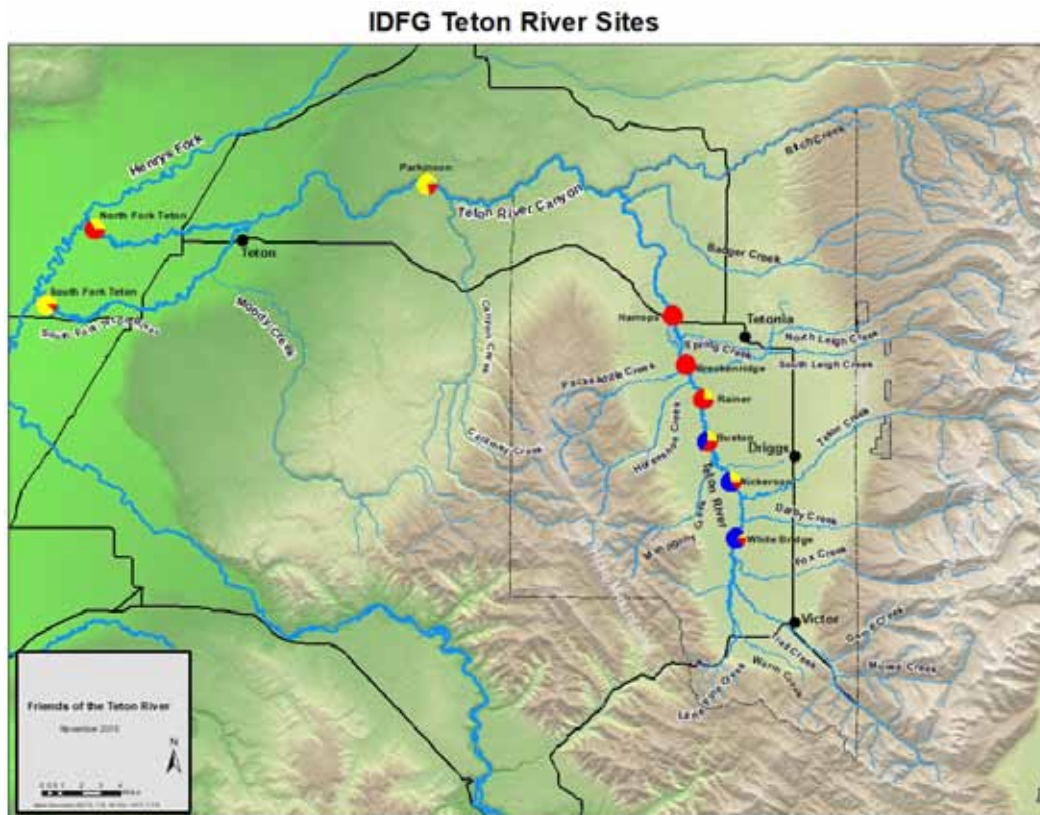


Figure 2. Idaho Department of Fish and Game electrofishing sites in the Teton River watershed.

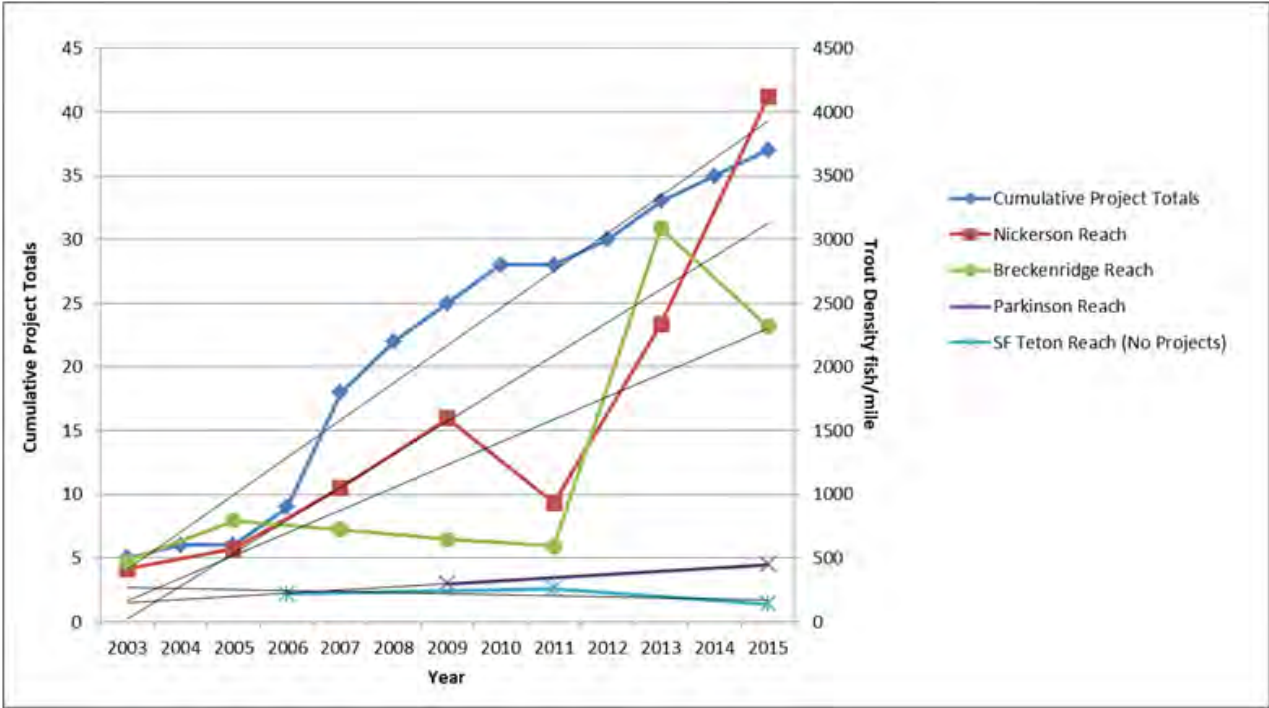


Figure 3. Changes in fish densities for all trout species in the Teton River compared to cumulative number of completed project totals. Best-fit regression lines are included for project totals and each sampling site.

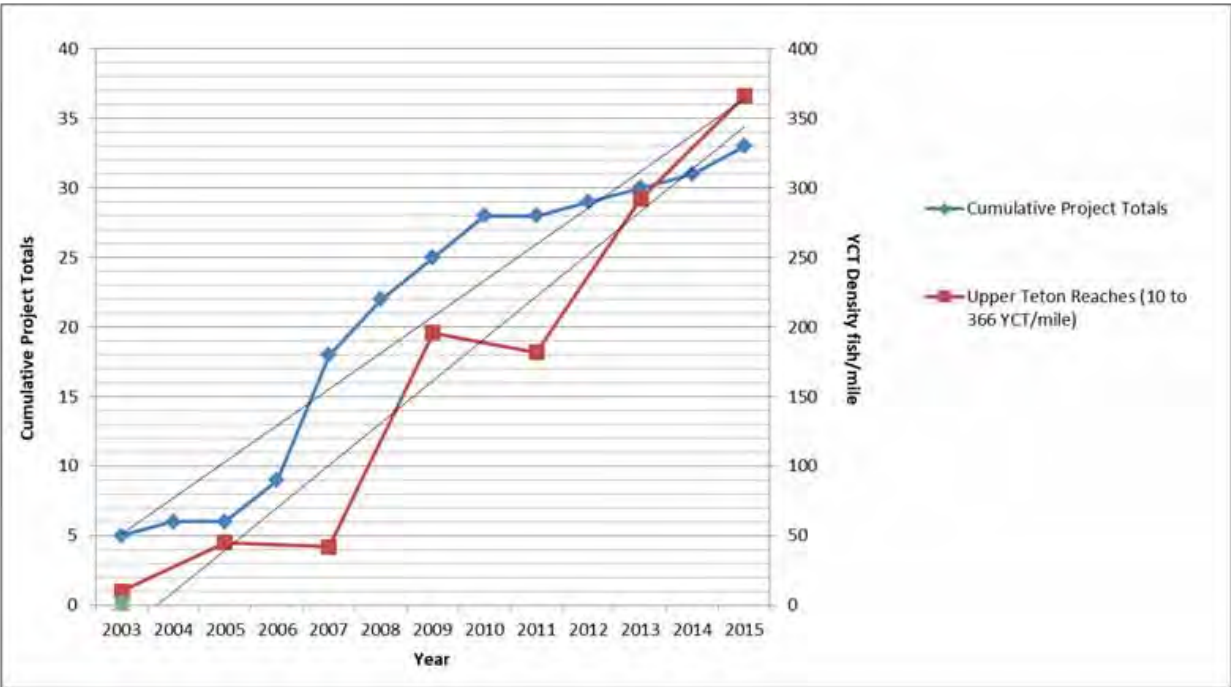


Figure 4. YCT densities in Upper Teton River for Nickerson and Breckenridge sites combined compared to the cumulative number of project totals. Best-fit regressions lines are included for project total and Upper Teton sampling reaches.

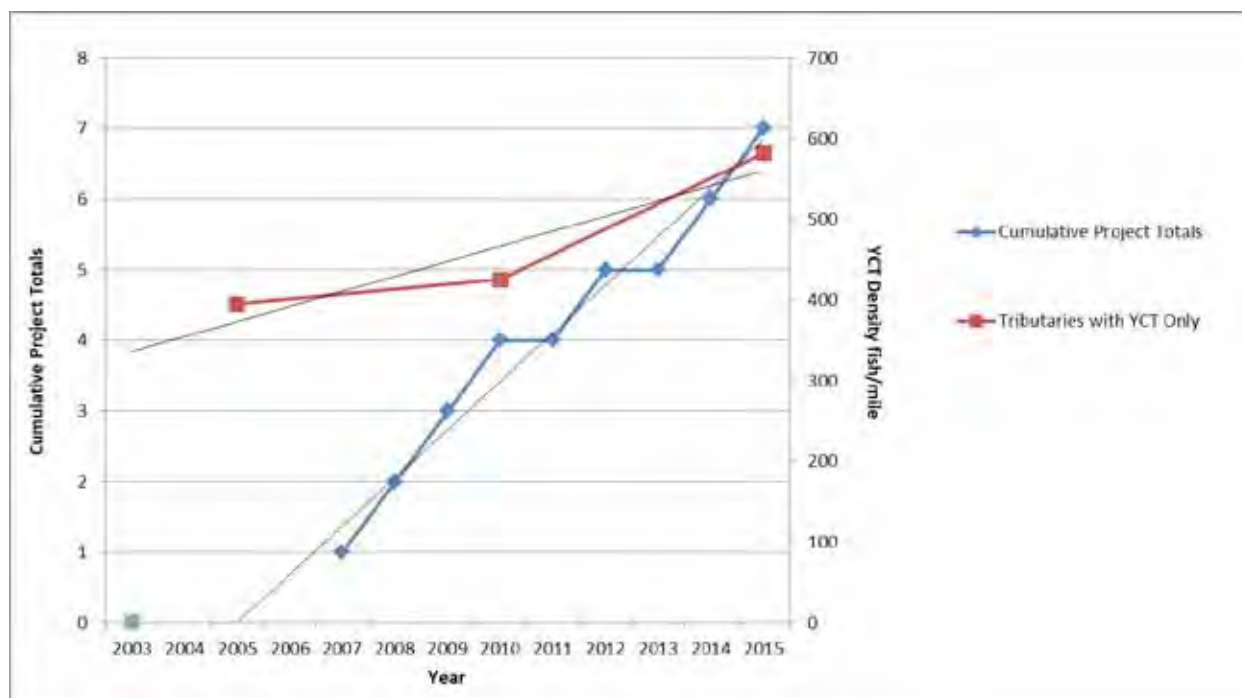


Figure 5. YCT Densities for Upper Teton River tributaries with only YCT present (Badger and South Leigh creeks) compared to the total cumulative number of project. Regression lines are included for project totals and YCT densities.



Identifying High-Value “Marginal” Brook Trout Populations Using a Conservation Portfolio Approach

Shawn Rummel¹, Kurt Fesenmyer², Amy Haak³, Matt Mayfield⁴, Mark Hudy⁵, and Jack Williams⁶

¹Field and Research Manager, Trout Unlimited, Lock Haven, PA, rummel@tu.org

²GIS and Conservation Planning Director, Trout Unlimited, Boise s, ID, kfesemyer@tu.org

³Executive Director, Conservation Geography, Boise, ID

⁴GIS Analyst, Trout Unlimited, Boise, ID

⁵Retired/Science Advisor, Eastern Brook Trout Joint Venture, Harrisonburg, VA

⁶Senior Scientist, Trout Unlimited, Medford, OR

Abstract—Brook Trout *Salvelinus fontinalis* (BT) are a species of conservation concern in the eastern U.S. Habitat degradation has led to widespread population decline and the occupation of marginal habitats. Marginal trout habitats can be defined based on population, habitat, or future threat factors and are traditionally thought of as those that support small, isolated populations, those that occur in degraded watersheds, or those that experience the extremes of a species' thermal tolerance. However, some marginal populations may be high value to the species' conservation and should be restoration priorities. The conservation portfolio approach evaluates BT populations to characterize their resiliency to disturbances, likelihood of demographic persistence, and representation of genetic, life history, and geographic diversity, as well as likely habitat condition and climate vulnerability using landscape-scale GIS datasets. The results of this project describe the extent and distribution of marginal BT populations, characterize marginal populations based on potential life history diversity, and examine the conservation value and strategies associated with marginal BT populations. This project offers a novel approach to identify priority areas for BT conservation and restoration and will aid decision makers in directing effort and funding to areas that will ensure the diversity and long-term viability of the BT across its native range.

“To keep every cog and wheel is the first principle of intelligent tinkering.”

--Aldo Leopold, Round River (1953)

Introduction

The Brook Trout *Salvelinus fontinalis* has experienced significant population declines throughout its native range. Brook Trout have been extirpated from 28% of historically occupied subwatersheds and greatly reduced (> 50% of populations lost) in another 35% of subwatersheds in the eastern United States (Hudy et al. 2008). The decline has been attributed primarily to anthropogenic activities that have degraded the water quality, habitat quality and diversity, and water temperature beyond the narrow tolerance ranges of Brook Trout (Hildebrand and Kershner 2000; Hudy et al. 2008; Stranko et al. 2008; Kanno et al. 2015; Wagner and Deweber 2015). These disturbances have primarily relegated Brook Trout to isolated populations in the headwaters of river networks (Hudy et al. 2008) where they are occupying suboptimal habitats.

Marginal habitats consist of resources and conditions beyond the species ecological niche (Hutchinson 1961; Pulliam 2000; Kawecki 2008). Some marginal populations are evolutionarily important due to local adaptations and genetic diversity (Kawecki 2008), such as those that occupy the periphery of a species range (Haak et al. 2010). For this project, we consider marginal Brook Trout populations from the perspective of population dynamics (low abundance and/or density; Hudy et al. 2008), habitat (as previously described), future threats to habitat condition (Deweber and Wagner 2015), or a combination of these factors (Fesenmyer et al. 2017). As Brook Trout populations continue to decline, the management of marginal populations is critical to ensure the survival and stability of the species. Particular considerations should be given to providing Brook Trout access to exceptionally productive

or diverse habitats so varied life histories can be expressed. The expression of diverse life histories confers and increases the stability and likelihood of persistence of populations (Rieman and Dunham 2000; Moore et al. 2014) and access to exceptional habitats allows for gene flow among previously isolated populations (Castric et al. 2001; Torterotot et al. 2014).

The goals of this project were to describe the extent and distribution of marginal Brook Trout populations, characterize marginal populations based on potential life history diversity, and examine the conservation value and strategies associated with marginal Brook Trout populations.

Methods

We used Eastern Brook Trout Joint Venture (EBTJV) Brook Trout population patches (Coombs and Nislow 2015) as our unit of analysis. EBTJV patch data map contiguous Brook Trout habitats in the form of a watershed boundary or population “patch” delineated based on presence/absence surveys and location of barriers such as dams. Patch attributes applicable to this analysis include overall size (area) and trout species composition information. We make an assumption that all stream habitats within the patch are occupied and accessible to Brook Trout, and use the EBTJV catchment-scale data to identify allopatric and sympatric Brook Trout stream habitats within patches.

Identifying and Mapping Marginal Brook Trout Populations

We surveyed published, peer-reviewed literature for descriptions of key change points associated with Brook Trout occurrence or abundance within studies using watershed characteristics. Where studies on Brook Trout were lacking, we substituted studies of other salmonid taxa. Once we identified the key population, habitat, and threat factors associated with marginal populations, we summarized spatial data within patches associated with each factor; EPA's StreamCat (Hill et al. 2016) served as our primary spatial data source, supplemented with other sources (EPA 2012; EPA and USGS 2005; DeWeber and Wagner 2015). We then calculated correlations among factors and removed any redundant variables. We summarized the pattern of each final factor within EBTJV patches identified as allopatric Brook Trout, sympatric Brook Trout and Brown Trout *Salmo trutta*

and/or Rainbow Trout *Oncorhynchus mykiss*, and for all populations at the 90th, 75th, and 50th percentile levels.

We applied published change point values to flag a patch as marginal except where multiple threshold values or a range of threshold values have been described. In those cases, we evaluated the median value within the EBTJV dataset to determine the threshold we applied in this study. Additionally, we refined the thresholds to ensure no less than 10% of the EBTJV patches met the criterion. We then mapped each population according to the thresholds for factors associated with population, habitat, and threat categories. If a population did not meet the threshold for any of the factors associated with the three categories, it was considered marginal. Final categories for each patch are marginal due to population factors, marginal due to habitat factors, and marginal due to threat factors. Populations can be marginal for one to three factors; populations not marginal for any factor are considered strongholds.

Mapping Life History Diversity

We characterized life history expression for each EBTJV population patch based on the diversity of habitats accessible to Brook Trout within patches or, in the case of anadromy, observed expression of a particular life history. Our habitat-focused characterization of populations assumes that individuals will develop life histories to exploit available habitats and that larger, more diverse or productive habitats lead to the expression of unique life histories. Various life histories and corresponding criteria applied in this analysis are described in Table 1. Primary data sources for characterizing stream and lake habitats are The Nature Conservancy's Northeast and Appalachian Aquatic Habitat Classification datasets (Anderson et al. 2013; Olivero Sheldon et al. 2015) and the National Hydrography Dataset (EPA and USGS 2005).

Evaluating Effect of Complete Barriers Dataset on Patch Delineation

To characterize how a complete barriers dataset can change the results of this analysis, we used a recent public dataset of culvert survey data from Maine (TNC 2017) to re-map Brook Trout population patches

Table 1: Criteria and rationale for life history categories. Stream size classes assigned to each reach based on NHD Plus contributing area - Class 1 (small headwaters): < 5 km²; Class 2 (headwaters): 5-10 km²; Class 3 (small creek): 10 -50 km²; Class 4 (creek): 50 – 100 km²; Class 5 (small river): 100 – 500 km²; Class 6 (river): > 500 km². Stream alkalinity classes based on The Nature Conservancy’s Northeast Aquatic Habitat Classification for EBT Northeast and Mid-Atlantic regions (NE) and Stream Classification for the Appalachian Region (App) for EBT Southeast region as follows: Class 1: (NE “Low (< 25 mg/L)”); App. “Low buffered, acidic”); Class 2: (NE “Medium (25 - 50 mg/L)”); App. “Moderately buffered, neutral and Assumed moderately buffered”); Class 3: (NE “High (50 – 150 mg/L) and Very high (≥ 150 mg/L:)”); App. “Highly buffered, calcareous”).

Life history	Criteria	Rationale
Migratory – river	Max. stream size class minus min. stream size class ≥ 2 & stream habitat ≥ 25 km OR length-weighted stream size class ≥ 3 & stream habitat ≥ 25 km	Diverse inter-connected stream habitats that include larger systems will support migratory life history
Migratory – lake	Lake size ≥ 0.25 ha & stream habitat ≥ 5 km OR lake size ≥ 5 ha & stream habitat ≥ 1 km	Connected lake and stream habitat of sufficient size will support migratory life history between lake and stream
Migratory – sea run (NE only)	Population identified in Dauwalter et al. 2014 as “High, Moderate, or “Low-moderate” certainty of anadromy	Anadromous life history expressed in coastal streams
Resident – more productive	Length-weighted stream size class ≥ 3 & length-weighted alkalinity class > 2	Larger and more alkaline streams are more productive and will support fast-growing and short-lived resident populations
Resident lake/pond (NE only)	Ponds or lakes not connected to NHD Plus stream network. Includes ME Heritage Waters or pond or NHD Plus lakes not characterized as “warm” or “hypereutrophic”	Isolated resident Brook Trout without access to stream habitats
Resident – less productive	All other populations	Small streams or lakes support less unique resident populations

following the methods of the original EBTJV patch delineation (Coombs and Nislow 2015). For each original EBTJV patch, we calculated a fragmentation index as the size of original patch divided by the size of the largest patch remaining within original patch. We report the average fragmentation index value by the original EBTJV patches after the re-patch exercise.

Results

Identifying and Mapping Marginal Brook Trout Populations

The key population, habitat, and threat factors and associated change points identified within peer-reviewed literature and a description of the 90th, 75th, and 50th percentile values associated with our final list of marginal population factors by patch were determined (Table 2).

Our candidate list for identifying marginal patches includes one population factor (amount of available stream habitat), six habitat factors (agricultural land use, watershed forest cover, riparian zone forest cover, road density, impervious surface, 303(d) listed streams), and one threat factor (August stream temperature). We eliminated impervious surface and percent forested land cover from our final list of factors associated with marginal populations after evaluating correlation among variables; impervious surface is correlated with road density ($r^2 = 0.75$) and percent forested land cover (watershed) is correlated with percent forested riparian zone ($r^2 = 0.97$).

Each factor is associated with a published threshold with the exception of 303(d) listed streams, which reflect water quality impairment. Although the presence of streams listed under Section 303(d) of the Clean Water Act has not, to our knowledge, been

Table 2: Percentile values associated with marginal population, habitat, and threat characteristics by Brook Trout patch, literature thresholds, and thresholds applied in this study. Applied thresholds marked with * are not included in this analysis due to correlation with other factors.

Percentile (patch count)	Allopatric patches			Sympatric patches			All patches			Literature Threshold	Applied Threshold
	90 th	75 th	50 th	90 th	75 th	50 th	90 th	75 th	50 th		
Population: Km stream habitat available	16.1	7.7	4.0	30.7	14.3	6.3	22.2	9.8	4.6	< 25 (Hilderbrand & Kerschner 2000)	< 20
Habitat: % agricultural land use	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	> 2.5 – 25 (Wagner & Midway 2014; DeWeber & Wagner 2015)	> 2.5
Habitat: % forested riparian zone	98	93	84	96	90	76	97	92	81	< 80 (as % forested watershed: Stranko et al 2008; Kanno et al. 2015)	< 80
Habitat: % forested watershed	96	91	80	95	88	75	96	90	79	< 80 (Stranko et al. 2008; Kanno et al. 2015)	*
Habitat: Road density (km/km ²)	0.4	0.9	1.4	0.6	1.0	1.5	0.4	0.9	1.5	> 2 (Hudy et al. 2008)	> 2
Habitat: Percent impervious surface	0.0	0.0	0.2	0.0	0.1	0.3	0.0	0.1	0.2	< 4 (Stranko et al. 2008)	*
Habitat: Km stream 303(d) listed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-	> 0
Threat: Ave. August stream temp. °C	16.5	17.1	17.8	16.8	17.4	18.2	16.6	17.2	17.9	> 18.5 – 21 (DeWeber & Wagner 2015)	> 18.5

evaluated as a predictor of Brook Trout occurrence or abundance, these streams do not meet basic water quality standards set forth in the Act. Furthermore, the median value (50th percentile) of allopatric, sympatric, and all Brook Trout populations is 0 km, reflecting the relative rarity of 303(d) listed streams within Brook Trout populations.

The EBTJV dataset (Coombs and Nislow 2015) identified a total of 9,811 allopatric and sympatric Brook Trout patches. Based on the framework described above, 9,566 (97.5%) of the EBTJV patches may be considered marginal by at least one of the factors; 2,393 (24.3%) patches are considered marginal due to the population factor alone, 568 (5.7%) due to habitat factors alone, and 24 (0.2%) due to the threat factor alone (Figure 1). The number of patches categorized as strongholds or marginal, along with the marginal factors described above and the life history

diversity are described in Table 3. Migratory lake life histories were primarily categorized as marginal by population and/or habitat factors (77% of total patches with this life history), while migratory river populations were primarily categorized as marginal due to habitat and threat factors (77% of total patches with this life history). Residential life histories considered marginal were influenced by all three factors.

Evaluating Effect of Complete Barriers Dataset on Patch Delineation

The EBTJV has identified a total of 72,323 road-stream crossings occurring within Brook Trout patches in the eastern U.S. (EBTJV 2015), with a mean of 7.3 road-stream crossings per patch (SD ± 14.5). The proportion of these crossings which function

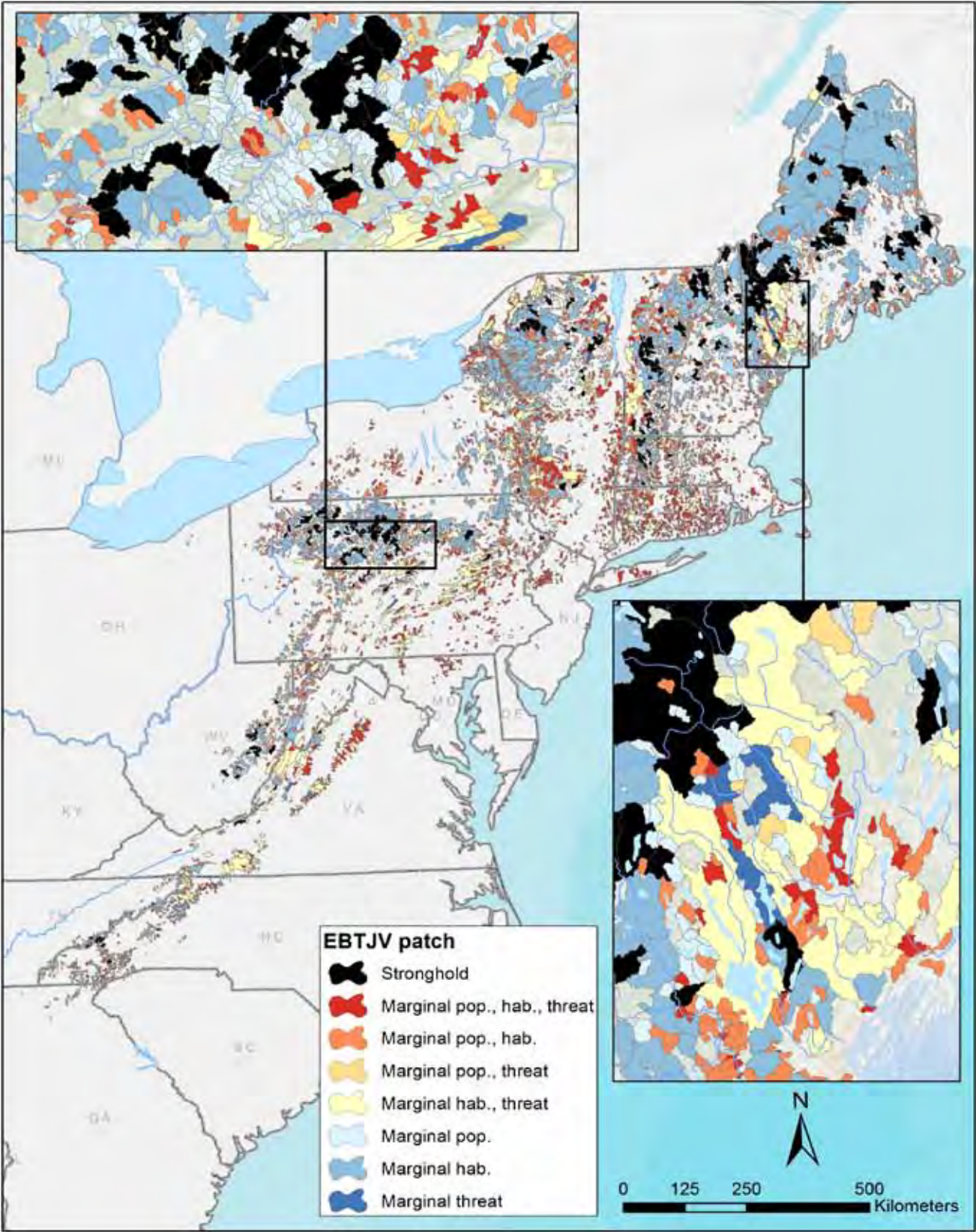


Figure 1: Classification of EBTJV Brook Trout population patches by marginal or stronghold categories.

as complete barriers to fish passage is unknown. To explore how a more comprehensive barrier survey can affect the size of delineated Brook Trout habitat patches, we calculated a fragmentation index from a recent barrier assessment in Maine. The average fragmentation index value for Maine ($n = 998$ patches) is 0.88 (range = 0.13 – 1.00), indicating that, on average, the largest patch internal to the original EBTJV patch after the re-patch exercise is 88.8% of the size of the original patch (Figure 2).

Discussion

Only 3% of Brook Trout patches in the eastern U.S. may be considered strongholds based on multiple published criteria for identifying marginal populations. Maintaining strongholds through protection and restoration efforts and creating new strongholds by removing barriers and connecting populations should be considered high conservation priorities (Fesenmyer et al. 2017). The remaining 97% of Brook Trout

patches may be considered marginal due to population, habitat, or future impairments, or a combination of the three factors (Table 3; Figure 1). Given the large number of patches that fall into this category, prioritization of conservation and restoration efforts is a necessity to focus efforts to “high value” patches and patches with the potential to provide the greatest return on conservation investment (Roni et al. 2010).

Life history expression provides a method to assign value to these marginal patches, with patches that express rare and unique life histories considered to be “high value”. Conservation strategies aimed at securing and protecting these high value populations should be implemented. Securing these populations may require restoration to address sources of impairment, protection actions to prevent new stressors from occurring, and mitigating future threats. Our characterization of life history expression was, with the exception of anadromy, based upon the habitats accessible to Brook Trout within a patch. Recent research has shown that Brook Trout will utilize habitats such as large rivers, lakes, ponds, etc. when they are available and are suitable (Curry et al. 2002; Roghair and Dolloff 2005; Petty et al. 2012). Seasonally-suitable habitat is critical to life history expression as larger main-stem rivers may not be available during the warm summer months due to increases in water temperature, but these areas provide habitat and potential migratory pathways during the cooler months of the year. Therefore, potential restoration opportunities should not be overlooked based on summer temperatures alone. Genetic diversity also contributes to the stability and resiliency of populations (Letcher et al. 2007) and may provide another method of assigning conservation value to marginal populations. However, a comprehensive, range-wide genetic dataset for Brook Trout is currently lacking.

Marginal populations that do not express rare and unique life histories may provide additional conservation opportunities for Brook Trout. Restoration may not be feasible or cost effective within some marginal patches; therefore, the complexity of restoration actions within a patch should be considered when prioritizing conservation work in marginal habitats. These data provide a framework to consider restoration complexity (Figure 1 and Table 3). Patches falling into the marginal category with a single factor (population, habitat, or threat) represent those with the lowest restoration complexity,

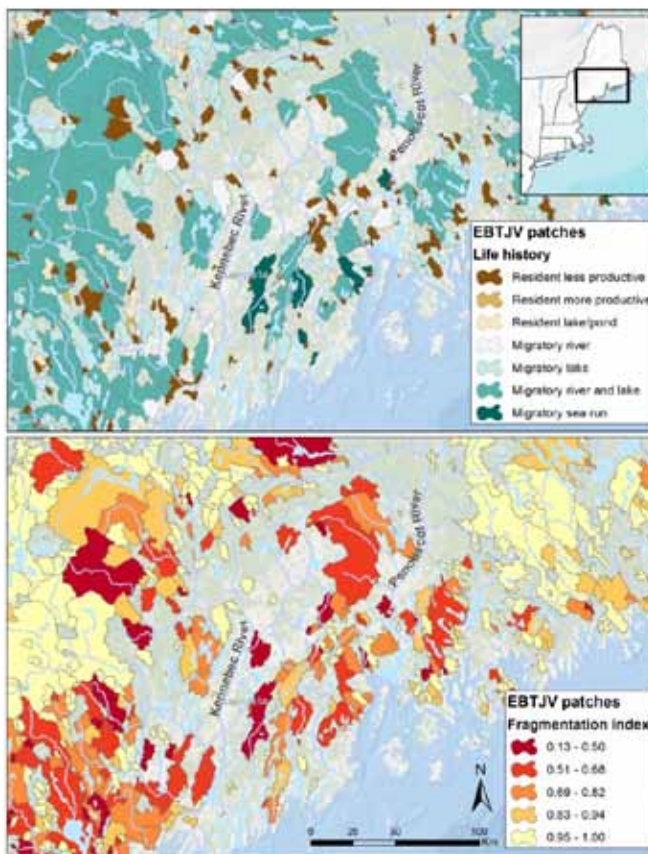


Figure 2: Example life history and fragmentation results for Maine.

Table 3: Count of stronghold and marginal populations by life history type. A small portion of patches are missing stream data (n = 148).

Life History	Total Count	Count Stronghold	Count Marginal						
			Pop. only	Pop. & Hab.	Pop. & Threat	Habitat only	Hab. & Threat	Threat only	Pop., Hab., & Threat
Migratory lake	790	20	265	343	30	43	6	2	81
Migratory river	458	92	0	0	0	202	150	14	0
Migratory river and lake	379	83	0	0	0	232	60	4	0
Migratory sea run	41	1	3	12	0	3	2	0	20
Resident lake/pond	265	1	110	104	17	3	1	0	29
Resident less productive	7,783	48	2,011	2,943	377	84	72	4	2,244
Resident more productive	95	0	4	19	0	1	1	0	70
Total	9,811	245	2,393	3,421	424	568	292	24	2,444

while those with all three factors likely represent the highest levels of complexity. Conservation strategies in these areas should focus on restoring impaired and vulnerable Brook Trout habitats to create new high value populations (i.e., increase patch size, increasing amount of available habitat with a patch, reconnecting adjacent populations, etc.). For example, marginal populations that are adjacent to strongholds may provide the opportunity for reconnection activities to create larger, more connected Brook Trout habitat.

The re-patch exercise described for Maine demonstrates how barrier assessment data may dramatically change the results of this analysis. Efforts are currently underway to assess fish passage barriers across the range of the Brook Trout (e.g., North Atlantic Aquatic Connectivity Collaborative); however, a complete dataset for the range is currently lacking. Once completed, this information may help refine our definition of marginal populations. There are several other factors that are missing from this analysis due to unavailable range-wide data that could refine our definition of marginal populations, inform our understanding of life history diversity, and be potential directions for future research. For the population factor, estimates of Brook Trout population size, observed life history, and characterization of the

genetic diversity of Brook Trout (including patterns of stocking and unique genetic lineages, such as potentially distinct Southern Appalachian Brook Trout (Hayes et al. 1996; Habera and Moore 2011)) would refine the definition of marginal populations. Additional habitat factors include the fish passage barrier assessments previously described and observationally-based data related to habitat condition. Broad scale models mapping vulnerability associated with increasing floods, drought, and changes in seasonality of stream flow (Melillo et al. 2014), or the lack of groundwater influence (Snyder et al. 2015) could be useful for refining the threat factor in marginal populations.

The results of this project outline conservation strategies that can be adopted to enhance the long-term viability of marginal populations as well as identify monitoring and research needs related to population, habitat, and future security attributes that would further our understanding of how “marginal” Brook Trout populations truly are. This project offers an approach to identify priority areas for Brook Trout conservation and restoration and will aid decision makers in directing effort and funding to areas that will ensure the diversity and long-term viability of the Brook Trout across its native range.

Acknowledgements

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Vulnerability of Gila Trout Streams to Future Wildfires and Temperature Warming

Daniel C. Dauwalter¹, Jack E. Williams², Joseph McGurrin³, James E. Brooks⁴, David L. Propst⁴

¹Trout Unlimited, 910 Main Street, Suite 342, Boise, Idaho 83702,

²Trout Unlimited, 4393 Pioneer Road, Medford, Oregon 97501,

³Trout Unlimited, 1777 Kent Street, Suite 100, Arlington, Virginia 22209

⁴Department of Biology, Museum of Southwestern Biology, University of New Mexico, Albuquerque, New Mexico 87131

Abstract—Climate change is projected to increase the frequency and severity of wildfires, warm stream temperatures, and negatively impact native trout habitat. Southwestern native trouts are often isolated from nonnative salmonids above conservation barriers and have a limited ability to recolonize after disturbances or move to track changing environmental conditions such as stream temperature. We combined wildfire, debris-flow, and 2080s stream temperature models to identify Gila Trout *Oncorhynchus gilae* habitats least vulnerable to these threats and guide conservation efforts. Wildfire risk, debris-flow probability, debris-flow volume, minimum 2080s mean August temperature, and kilometers of habitat with 2080s August temperatures <18.5°C were summarized for each Gila Trout stream and ranked for overall vulnerability. The vulnerability rankings can be used to inform conservation actions such as reintroductions, habitat restoration, or nonnative fish eradications while considering these climate-related threats and other factors. Conservation decisions mindful of climate resiliency will best ensure that these unique but threatened native trouts remain on the landscape in the southwestern U.S. in future climates.

Introduction

The Earth's climate is changing. The last two years, 2015 and 2016, were the warmest on record. Carbon dioxide levels in the atmosphere are higher than ever recorded. In the western United States, mountain snowpack is melting earlier, and earlier and drier springs are increasing the frequency and intensity of wildfires (Westerling et al. 2006). Streamflows during dry years are getting lower (Luce and Holden 2009), and stream and river temperatures have been increasing 0.3°C per decade (Isaak et al. 2012).

Stream salmonids can be vulnerable to wildfires (Dunham et al. 2003). Wildfires can superheat stream water, cause ash flows that alter water quality, or result in channel reorganizing debris flows, each of which can cause direct mortality (Gresswell 1999; Cannon et al. 2010). Salmonid populations with limited dispersal ability can be particularly vulnerable to wildfire due to an inability to recolonize fire-impacted habitats (Dunham et al. 2003). As ectotherms, salmonids are also particularly vulnerable to climate warming, although climate change is expected to affect

populations through streamflow alteration and drought in addition to temperatures (Williams et al. 2009; Wenger et al. 2011).

In the southwestern United States, the climate is projected to warm rapidly over the next century. Kennedy et al. (2009) used a regional climate model and projected summer air temperatures to increase 2°C and a 20% decrease in precipitation by the 2050's. They used these projections to estimate that Gila Trout *Oncorhynchus gilae* habitat would decrease by 70% in that same time frame, albeit using air temperature as a surrogate for stream temperature. Concomitant decreases in humidity and more frequent drought conditions are also likely to accompany changes in temperature, thus resulting in larger and more intense wildfires (Williams and Carter 2009).

Catastrophic wildfires and number of extant populations were major factors influencing the viability of Gila Trout in the southwestern United States (Brown et al. 2001). Like other salmonids, southwestern native trouts have narrow physiological tolerances, especially thermal tolerances (Lee

and Rinne 1980; Recsetar et al. 2013). Because populations are typically isolated above conservation barriers (Propst and Stefferud 1992), southwestern trouts have limited ability to respond to environmental changes through movement or recolonization and, therefore, have a high vulnerability to changes in climate (Rinne 1982; Kennedy et al. 2009).

Our objectives were to (1) summarize wildfire history within the historical range of Gila Trout, and (2) use spatially explicit wildfire, debris flow, and stream temperature models to identify Gila Trout streams least vulnerable to these threats and inform conservation efforts. The Gila Trout is listed as Threatened under the U.S. Endangered Species Act (prior to 2006 the species was listed as Endangered).

Methods

Wildfire History

We summarized fire history in New Mexico and Arizona within the historical range of Gila Trout in the southwestern United States that includes the Gila, Salt, and Verde river systems (Benke 2002). The historical range of Gila Trout surrounds the historical range of Apache Trout *O. apache* in the Black and White rivers in the headwaters of the Salt River, and we summarized fire history within this region as well. We used the Monitoring Trends in Burn Severity program database to summarize fire frequency, fire extent, and ignition timing from 1985 to 2015, the most recent year available (Eidenshink et al. 2007).

Wildfire Risk

We developed spatially-explicit estimates of wildfire risk for Gila Trout streams using FlamMap 5.0 software. FlamMap models fire behavior characteristics from a static set of environmental conditions: fuel moisture based on vegetation type, wind speed and direction, and topography. FlamMap models active and passive crown fire potential using weather conditions, including wind interactions with topography, and we used crown fire potential as a measure of wildfire risk. We used WindNinja software to model wind routing through the landscape and initialize wildfire behavior for input into FlamMap (Forthofer 2007). To parameterize WindNinja, we used average daily maximum wind gust speed and average wind direction using data during the fire season (April 1 through August 31; see Results) from 2010 to 2015

as summarized from six Remote Automated Weather Station (RAWS) stations representative of our study area: Greer (AZ), Mountain Lion (AZ), Alpine (AZ), Beaverhead (NM), Mogollon (NM), and Pelona Mountain (NM) (<http://www.raws.dri.edu/>). We used an average maximum wind speed of 24 km/h (6.1 m above ground) based on observed wind speeds, and we modeled wind routing as a weighted-average of the proportion of average daily wind directions at 16 azimuthal directions (20°, 40°, 60°, 90°, 120°, 140°, 160°, 180°, 200°, 220°, 240°, 270°, 300°, 320°, 340°, 360°) across the six RAWS stations. Wind routing was implemented in WindNinja based on interactions with landscape topography (slope and aspect) from a 30-m digital elevation model. The most recent vegetation data from 2014 (includes 2014 fire season) were acquired from LANDFIRE (<http://www.landfire.gov>) and used as fire fuel input (Stratton 2009). Fire fuels were based on the 40 Scott and Burgan Fire Behavior Fuel Models, which represents fuel loadings based on vegetation types, size classes, and other fuels (Scott and Burgan 2005). Default fuel moisture levels were used for each vegetation type.

The spatial predictions of active and passive crown fires from FlamMap were summarized within the watershed upstream of all stream segments in the study area using the National Hydrography Dataset Plus (NHD+) version 2. The NHD+ dataset represents 1:100,000 map scale hydrography for all confluence-to-confluence stream segments; NHD+ stream segments average approximately 1-km in length. Wildfire risk was expressed as the percentage of each watershed predicted to have active or passive crown fire.

Debris Flow Risk

Wildfire risk and other physiographic factors were used to model post-fire debris flow probability and debris flow sediment volume (if a debris flow were to occur) using models from Cannon et al. (2010). Post-fire debris flow probability was computed as: $P_{\text{debris flow}} = e^x / 1 + e^x$, where: $x = -0.7 + 0.03 \cdot \text{BG30} - 1.6 \cdot \text{Rugg} + 0.06 \cdot \text{HSBurn} + 0.2 \cdot \text{Clay} - 0.4 \cdot \text{LiqLim} + 0.07 \cdot \text{StormInt}$, and: BG30 is the percent watershed area with slopes greater than 30%; Rugg is the watershed ruggedness computed as watershed relief (elevation maximum – minimum) divided by square-root of watershed area; HSBurn is the percent of watershed area burned at moderate to high burn severity (here replaced with percent watershed area

predicted to have active or passive crown fire as described above); Clay is the average clay content of soil in watershed; LiqLim is the average liquid limit of soils in watershed; and StormInt is the average storm rainfall intensity (mm/h) in the watershed (replaced with average 30-min storm intensity at a 2-year recurrence interval mm/h from National Weather Service). Watershed characteristics were computed using geospatial datasets as described in Cannon et al. (2010).

The predicted volume of debris flow material (V units: m³) was: $\ln(V) = 7.2 + 0.6 \cdot (BG30) + 0.7 \cdot HSBurn^{0.5} + 0.2 \cdot TotStorm^{0.5} + 0.3$, where: BG30 is as defined above; HSBurn is as defined above (also replaced with percent watershed area predicted to have active or passive crown fire); TotStorm is the total storm rainfall in watershed (mm) (replaced with average 30-min storm intensity [mm/h] at a 2-year recurrence interval) (Cannon et al. 2010).

Debris flow probabilities and volumes were modeled for all segments in the NHD+ dataset in our study area. Thus, each ~1-km stream segment has a debris flow probability and volume that reflects wildfire risk and other watershed characteristics.

2080s temperature risk

We evaluated stream temperature risk to climate warming using stream temperature models developed for New Mexico and Arizona. These models predict mean August temperatures measured *in situ* using digital thermographs as a function of elevation, canopy cover, stream slope, precipitation, drainage area, latitude, lakes and reservoirs, groundwater influence, air temperatures, and streamflows using a spatial statistical modeling approach (Isaak et al. 2016). Temperature projections for the 2080s were based on August air temperature inputs from a global climate model ensemble for the A1B warming trajectory. Model details can be found at: www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html. The New Mexico model was fit using 755 site-years of data, and the Arizona model was fit using 251 site-years of data. The New Mexico model had a root mean squared prediction error (RMSPE) of 1.03°C and the Arizona model had a RMSPE of 1.06°C, each suggesting the mean August temperature predictions were accurate to within ~1°C 66% and ~2°C 95% of the time. The models were used to make spatially explicit mean August temperature predictions for 1-km stream segments in the study area using NHD+ stream segments.

Gila Trout Stream Vulnerability

We summarized wildfire risk, debris flow risk, and 2080s stream temperature risk for all Gila Trout streams identified as potentially being useful for conservation. Gila Trout stream extents were delineated using a combination of field data and professional judgement. Streams were classified as: current population; recently restored population, recently extirpated (due to recent fires), and potential recovery stream. For each stream we summarized the average percent wildfire risk, mean debris flow probability, mean debris flow volume, minimum mean August stream temperature projected for the 2080s, and the kilometers of each delineated stream projected to have mean August temperatures below 18.5°C in the 2080s. These summaries were completed for each of 57 Gila Trout streams (or stream segments) identified for conservation purposes within the historical range of the species. All stream averages were length (habitat extent) or area (watershed) weighted. The temperature 18.5°C was based on the 95th percentile of all temperatures (averaged from 2002 to 2011) within Gila Trout streams classified as having a current, recently extirpated, or recently restored population where mean August temperatures were presumably suitable.

Results

We summarized wildfire, debris flow, and 2080s stream temperature risk information and vulnerability for 57 Gila Trout streams or stream segments in New Mexico and Arizona that represent 14 current populations, three recently restored populations, and four recently extirpated populations, as well as 25 streams identified as having potential for species reintroductions (Figure 1; Table 1).

Wildfire History

Within the broad historical range of Gila Trout in New Mexico and Arizona, there were 272 fires from 1985 to 2015, and 238 of those were wildfires totaling over 1.3 million ha (top left panel of Figure 2). Wildfires started during all months of the year, but a majority started in June at the onset of the monsoon season (top right panel of Figure 2). The median fire size from 1985 to 2015 was 1,300 ha, with an increasing trend in the maximum fire size and total area burned over time that reflects the recent and large catastrophic Rodeo (2002), Wallow (2011),

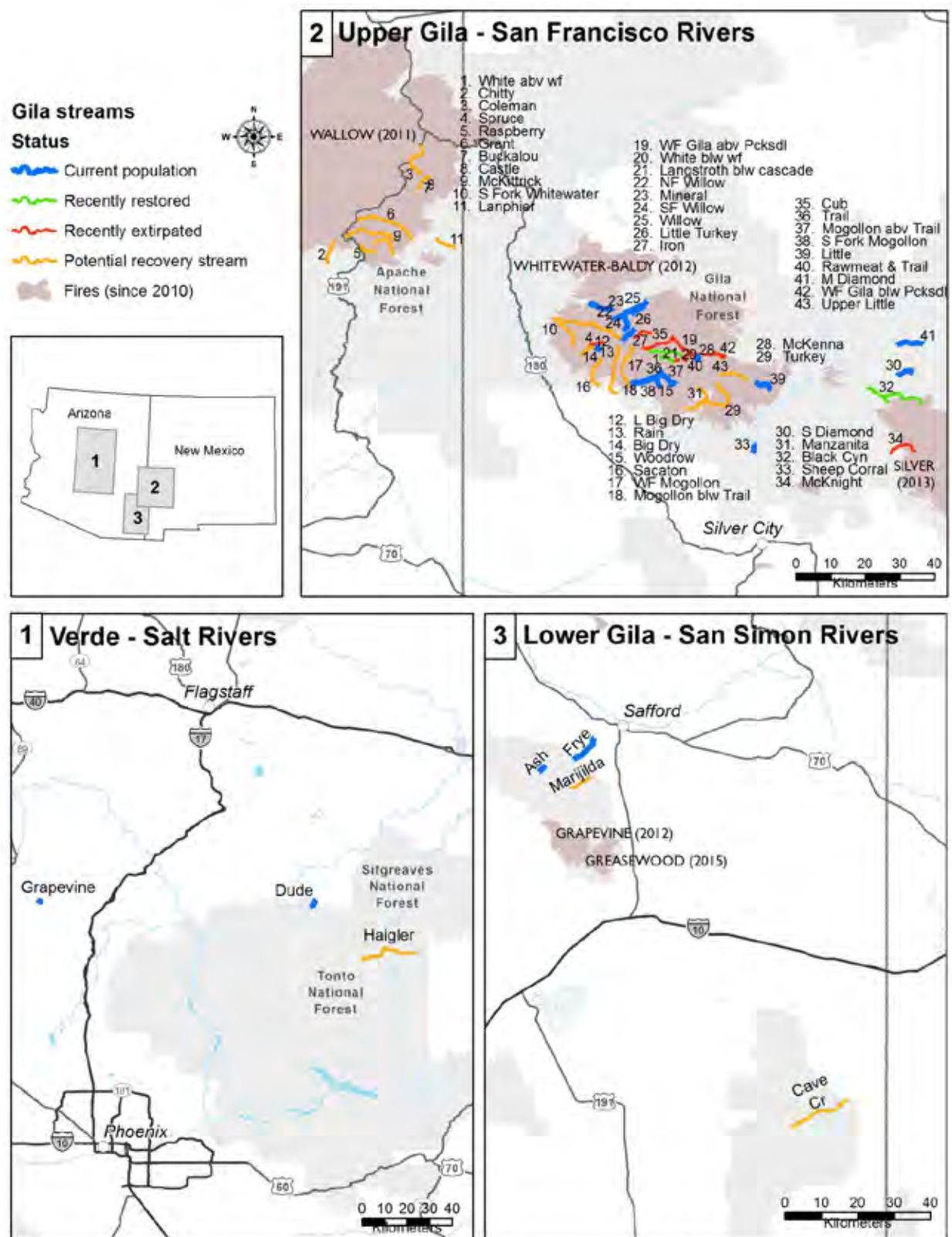


Figure 1. Gila Trout streams by population status in New Mexico and Arizona. Fires from 2010 to 2015 shown.

Table 1. Percent watershed with high wildfire risk (active or passive crown fire), mean debris flow probability given wildfire risk, debris flow volume, minimum mean August temperature in the 2080s, and habitat extent (km) below 18.5°C in the 2080s, and overall vulnerability rank of Gila Trout streams (a rank of 1 being least vulnerable).

Status	Population	Stream	Crown Fire (%)	Debris Flow Probability	Debris Flow Vol. (1000s m ³)	Min. °C (2080s)	Km <18.5°C (2080s)	Vulnerability Rank
Current	Ash	Ash	16.7	0.027	16	11.2	3	18
	Big Dry	Big Dry	41.7	0.003	20	12.6	2	20
	Dude	Dude	--	--	--	16.3	3	53*
	Frye	Frye	1.8	0.002	7	21.2	0	26
	Grapevine	Grapevine	--	--	--	17.4	2	55*
	Iron	Iron	25.7	0.008	12	10.2	4	16
	Little	Little	50.2	0.026	713	18.8	0	52
	Main Diamond	Main Diamond	60.0	0.238	234	14.9	10	45
	McKenna	McKenna	21.6	0.011	192	16.6	2	40
	Mineral	Mineral	37.9	0.005	44	14.7	9	22
	Mogollon	Mogollon above Trail	49.5	0.088	194	16.7	5	48
		Mogollon below Trail	58.2	0.020	983	17.6	9	46.5
		South Fork Mogollon	45.6	0.004	43	16.5	4	34
		Trail	62.9	0.006	20	15.4	4	32
	Sheep Corral	Sheep Corral	51.7	0.007	26	18.2	1	44
	South Diamond	South Diamond	60.0	0.015	96	13.9	11	31
	Willow	Little Turkey	3.1	0.001	10	13.5	7	1
		NF Willow	21.4	0.002	6	11.9	4	7
		SF Willow	26.0	0.003	9	11.2	7	6
		Willow	8.1	0.004	37	14.3	8	14.5
Eliminated	McKnight	McKnight	51.2	0.008	41	12.8	10	23
	Spruce	Spruce	49.1	0.024	20	12.1	4	29.5
	West Fork Gila	Cub	20.8	0.006	18	11.7	10	5
		Langstroth below cascade	44.9	0.019	30	15.9	2	38
		Rawmeat & Trail	23.4	0.016	33	16.0	2	35
		WF Gila above Packsaddle	24.6	0.026	447	16.5	6	39
		WF Gila below Packsaddle	42.7	0.023	9927	17.4	12	37
		White below waterfall	32.1	0.009	274	16.1	3	41
	Whiskey	Whiskey	14.7	0.011	13	11.1	5	8
Recovery		Buckalou	25.5	0.001	3	15.4	4	14.5
		Castle	33.9	0.001	32	16.8	3	28
		Cave Creek	--	--	--	11.1	9	42*
		Chitty	15.6	0.006	27	13.1	8	12.5
		Coleman	12.0	<0.001	32	14.6	15	2
		Grant	23.0	0.002	33	12.9	17	3
		Grant (Low)	54.4	0.005	165	18.5	1	51
		Haigler	--	--	--	18.3	2	56*
		Haigler (Low)	--	--	--	19.3	0	57*
		KP	23.2	0.003	59	13.0	15	9.5
		Lanphier	39.9	0.005	51	14.9	8	27
		Lower Big Dry	49.9	0.012	82	14.7	8	33
		Manzanita	63.3	0.248	59	15.7	6	46.5
		Marijilda	30.0	0.115	86	14.9	5	36
		McKittrick	30.9	0.002	23	13.8	8	12.5
		Rain	59.2	0.002	22	10.2	9	11
		Rain (Low)	66.7	0.003	145	18.2	2	50
		Raspberry	20.4	0.002	11	14.5	9	4
		Sacaton	58.2	0.003	30	11.3	9	17
		South Fork Whitewater	44.8	0.055	43	12.3	12	24.5
		Turkey	59.4	0.144	85	15.4	9	43
		Turkey (Low)	68.0	0.081	643	19.7	0	54
		Upper Little	55.5	0.169	147	17.4	8	49
		West Fork Mogollon	55.0	0.006	86	12.7	12	24.5
		Whitewater	48.0	0.018	96	9.8	20	19
Restored	Black Canyon	Black Canyon	48.9	0.007	409	11.7	25	21
	West Fork Gila	Langstroth above cascade	36.8	0.015	25	14.3	4	29.5
	White	White above waterfall	21.4	0.004	26	13.6	9	9.5

*High overall ranking due to no fire risk or debris flow data.

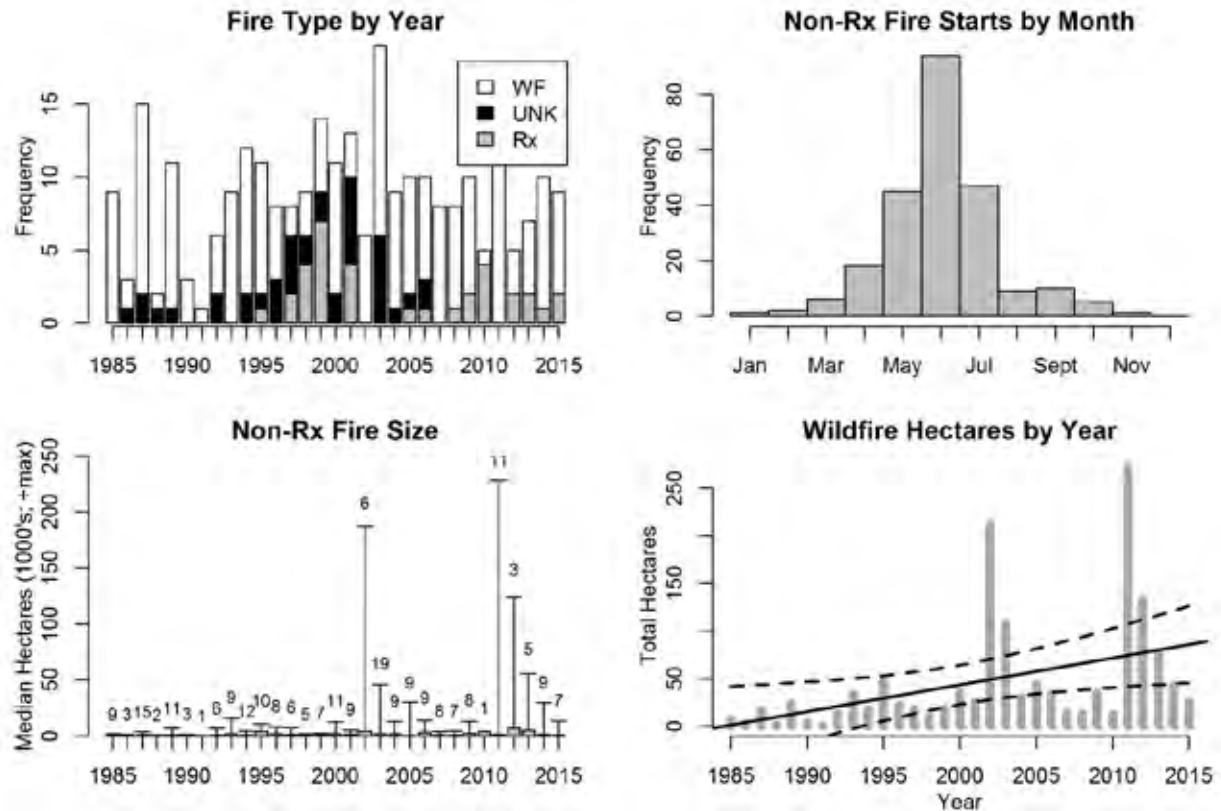


Figure 2. Frequency of wildfire (WF), prescription (Rx), and unknown (UNK) fire types by year (top left panel), frequency of fire starts by month (top right panel), median fire size (error bars = maximum; number of fires above bar) by year (bottom left), and total hectares burned by wildfire by year with trend line and 95% confidence intervals ($b_{\text{Year}} = 2.82$; $df = 29$; $P = 0.019$).

Whitewater-Baldy (2012), and Silver (2013) fires (Figure 1; bottom panels of Figure 2).

Wildfire Risk

Gila Trout streams exhibited a wide range of wildfire risk. The percent of watershed with high wildfire risk (active and passive crown fires) ranged from 2% in Frye Creek to 63% in Trail Creek for current and recently restored populations, and from 12% (Coleman) to 68% (Lower Turkey) for potential recovery streams (Table 1). Streams with Gila Trout populations extirpated by recent fires still had from 15% (Whiskey) to 51% (McKnight) of their watershed with high wildfire risk. Not surprisingly, wildfire risk was low within old burn perimeters where burn severity was highest (not shown), such as at high elevations on Mount Baldy within the 2012 Whitewater – Baldy fire perimeter (top panel of Figure 3).

Debris Flow Risk

The risk of post-fire debris flows was generally low in Gila Trout streams (Table 1). Probabilities of a debris flow occurring given modeled wildfire risk and other physiographic factors within the watersheds ranged from 0.001 or less (multiple streams) to 0.25 (Manzanita; Table 1). Watersheds with higher debris flow probabilities were clustered in certain drainages, such as the Turkey – Manzanita drainage in the eastern Mogollon Mountains (Figure 1; middle panel of Figure 2). Predicted debris flow volumes, if a debris flow were to occur, ranged from 3,000 m³ (Buckalou) to nearly 10 million m³ (lower West Fork Gila River). The Gila Trout streams with the highest probability of a debris flow generally had a moderate predicted debris flow volume; likewise, streams with the highest predicted volumes generally had less than a 3% chance of a debris flow occurring (top panel of Figure 4).

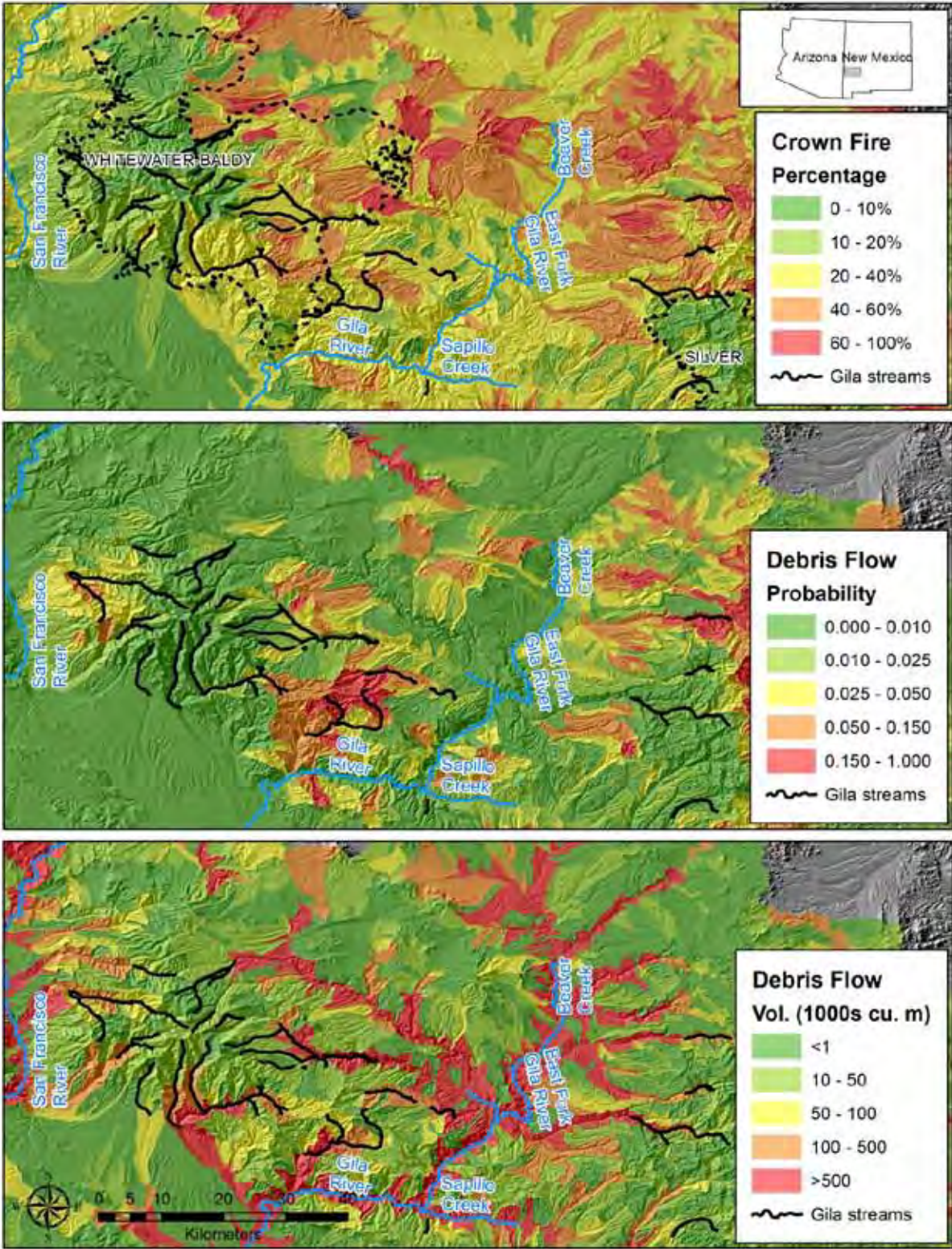


Figure 3. Example of percent watershed with high wildfire risk (active or passive crown fire) from FlamMap model (top panel), predicted debris flow probability given wildfire risk in watershed (middle panel), and predicted debris flow volume (bottom panel) for Gila Trout streams in New Mexico. Whitewater – Baldy and Silver fire perimeters shown in top panel (black dashed line).

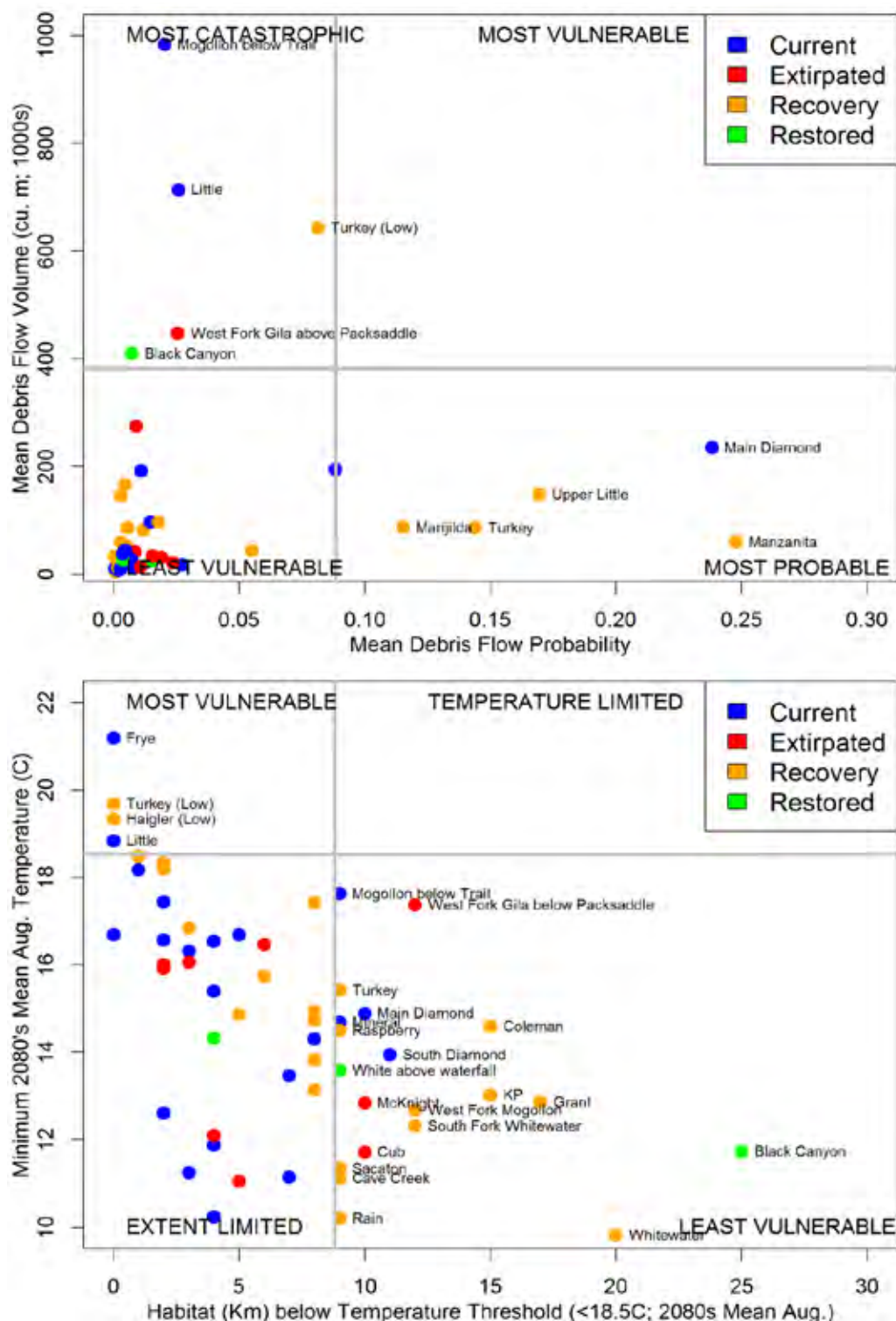


Figure 4. Mean predicted debris flow probability versus mean predicted debris flow volume (top panel) for Gila Trout streams, and length of habitat predicted to have mean August temperatures $<18.5^{\circ}\text{C}$ in the 2080s versus the minimum predicted mean August temperature in the 2080s across all stream segments (bottom panel) per Gila Trout stream. Streams symbolized by current population, recently restored population, population recently extirpated, and potential recovery stream.

2080s Remperature Risk

Gila Trout streams had varying risk to climate warming. Some streams had 0 km projected to have 2080s mean August stream temperatures below 18.5°C (Frye, Lower Turkey, Haigler, and Little), a temperature threshold that represents 95% of stream segments currently occupied by extant Gila Trout populations (bottom panel of Figure 4). Other streams have minimum projected 2080s mean August temperatures that are barely below that threshold. In contrast, a few potential recovery streams, such as Whitewater Creek, have large extents of habitat below that threshold and have minimum August temperatures projected to be less than 12°C in the 2080s. In fact, many streams had at least 8.8 km of habitat below 18.5°C, which is a habitat extent threshold commonly associated with a high likelihood of trout population persistence (Haak and Williams 2012).

Gila Trout Stream Vulnerability

When the 57 Gila Trout streams were ranked according to the five wildfire, debris flow, and temperature risk factors, a mix of potential recovery streams and current populations portended their low vulnerability to wildfire and 2080s temperature increases due to climate change. Little Turkey Creek ranked as the least vulnerable (overall rank = 1; Table 1). Together, the Willow Creek system containing Little Turkey, North Fork Willow, and South Fork Willow creeks appears to represent a stream network resilient to future wildfires, post-wildfire impacts (debris flows), and projected impacts of climate warming. Coleman and Grant creeks were also potential recovery streams that ranked in the top five for being least vulnerable. Interestingly, Cub Creek and Whiskey Creek now have low vulnerability despite recently being extirpated due to impacts from the Whitewater – Baldy Fire in 2012. This likely reflects the change in post-fire vegetation and fuels that are now not conducive to crown fires.

Discussion

Efficient conservation requires strategic investments of resources. Increasingly, climate change is playing a larger role in natural resource planning, as wildfires become more intense, streamflows decline, and stream temperatures warm (Williams et al. 2009; Isaak et al. 2015). We used wildfire, debris flow, and

stream temperature models to identify Gila Trout streams least vulnerable to these future threats for use in planning conservation actions.

Establishing additional viable populations within the historical range has long been a goal of the recovery plan for Gila Trout, including replication of the different genetic lineages (USFWS 2003; Wares et al. 2004). Our analysis suggests that several recovery streams already identified as potential reintroduction streams are likely to be least vulnerable to future wildfires and projected changes to stream temperatures. For example, Grant and Coleman creeks showed low risk to wildfire impacts and had over 8 km of habitat with suitable stream temperatures in the 2080s. Conservation efforts focused on recovery streams should consider their vulnerability to future wildfire and climate warming impacts.

Gila Trout streams with moderate vulnerability rankings may benefit from strategic restoration efforts to offset wildfire impacts and climate warming. The wildfire, debris flow, and temperature models all have elements representing landscape features that can be influenced by management. For example, forest vegetation can be managed to promote forest health and reduce wildfire severity in drainages with high potential for crown fires (Gresswell 1999). Watersheds with higher susceptibility to debris flows could receive high priority for post-fire revegetation efforts (Cannon et al. 2010). Restoration actions that reconnect streams to floodplains, restore riparian areas, and improve instream habitat all have the potential to buffer the impacts of climate warming on stream temperatures (Williams et al. 2015).

Others have built fire risk and future temperature predictions into decision support tools for native trouts. Falke et al. (2015) developed a vulnerability assessment for Bull Trout *Salvelinus confluentus* in the Wenatchee River system, Washington, under current and future climate scenarios that account for wildfire risk, projected changes to stream temperatures, and other factors. The assessment was used to evaluate different forest vegetation, riverine connectivity, and nonnative species management scenarios to determine where and what types of management would best benefit Bull Trout persistence in the watershed. The wildfire, debris flow, and temperature models we developed herein could similarly be integrated into a decision support tool to guide conservation actions. Such a tool should not only account for the climate

related factors described herein but also include factors such as streamflows, instream habitat, nonnative species to formally and completely compare and contrast Gila Trout conservation actions and guide decision-making.

Southwestern trouts are in peril. Both Gila Trout and Apache Trout have a limited number of extant populations that occupy a small portion of their historical range and each species is listed as Threatened under the Endangered Species Act. Strategic conservation will not only require the replication of genetic lineages and elimination of threats from nonnative salmonids, but it should also consider the vulnerability of native trout habitats to future wildfire and climate warming, among other factors, to ensure conservation actions across the landscape are climate resilient and long lasting.

Supplementary Materials

Available: <http://www.tu.org/gila-vulnerability>

Acknowledgments

We thank M. Mayfield for implementation of the fire and debris flow models and help with several figures. This project was funded by the National Fish and Wildlife Foundation and Trout Unlimited's Coldwater Conservation Fund.

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Restoring Brown Trout in the Vindel River in Northern Sweden

Daniel Palm, Göran Spong, Jan Nilsson, Anders Wiren, Helena Königsson,
Annika Holmgren and Daniel Jonsson

Swedish University of Agricultural Sciences, Dept. Wildlife, Fish and Environmental Studies, 901 83 Umeå, Sweden.

Extended Abstract

Introduction

Native populations of salmonids have declined worldwide due to human impacts on stream ecosystems. Important factors for the decline in stocks are overfishing, migratory barriers, and reduction in habitat. In northern Europe, declining stocks have historically been mitigated by stocking of hatchery reared juveniles that have been adjusted to rearing environments and pelleted food during several years. However, ambiguous stocking programs have commonly failed whereby stocking has been abandoned and is currently often regarded as a taboo among fishery managers. During the last decades, stock restoration has focused on solving problems related to poor connectivity and habitat quality and availability. Furthermore, large efforts to reduce harvest pressure from the commercial and recreational fisheries have also been conducted. Nevertheless, despite decades of these actions, whole catchments, or parts of, have had poor development of salmonid density. The Vindel River in northern Europe is one typical example. The main stem and its tributaries were severely damaged by channelization for timber floating and production of hydropower during the 19th and 20th centuries. Following 50 years of unsuccessful stocking, a large-scale nursery area and connectivity restoration program started in the 1990's. The restoration program was assumed to result in a quick recovery of native Brown Trout *Salmo trutta* L. Following modification of a fish passage system in the main stem and declining pressure from commercial fisheries in the sea, the number of ascending anadromous individuals showed a small increase. Nonetheless, despite two decades of physical habitat restoration many parts of the catchment are still poorly or not colonized by trout. In 2015 the river management organization decided to investigate if stocking of trout, according to a different methodology than previously used, could be used as a complementary tool to restore a trout population in parts of the catchment. The river management organization decided to apply stocking of eyed eggs

which would allow fry to emerge and adjust naturally to the stream environment. In order to evaluate the stocking, a parallel research program was started that would operate from 2015 through 2021. The aim of the research program is to study survival and dispersal of stocked individuals. The aim is also to genetically map the catchment yet not disturb unique native subpopulations.

Study Area

This study was conducted in the Vindel River in northern Sweden (Figure 1). The river arises in the Scandinavian mountain range and flows in a southeasterly direction and ends up merging with the Ume River approximately 42 km upstream where it drains into the Gulf of Bothnia on the east coast of Sweden. The water flow follows a snowmelt-dominated flow regime. The maximum discharge occurs in June and is measured up to approximately $1000 \text{ m}^3 \cdot \text{s}^{-1}$. Average annual flow is $180 \text{ m}^3 \cdot \text{s}^{-1}$. The river supports several subpopulations of trout. Both anadromous and resident as well as potamodromous populations occurs. Several other species are also found e.g. Atlantic salmon *Salmo salar* L., European grayling *Thymallus thymallus* L., northern pike *Esox Lucius* L., Eurasian perch *Perca fluviatilis* L., Euroasian minnow *Phoxinus phoxinus* L. and burbot *Lota lota* L.

Methods

By sampling tissue from trout in all main tributaries and at several locations along the main stem and quantifying Single Nucleotide Polymorphism (SNP) variation for 96 markers, the genetic relationship between subpopulations is possible to detect. Stocking of eggs has so far been conducted in two tributaries, one of which hosts a sparse population of trout and one completely lacks trout. Stocking was conducted in mid-April. Snow and ice was removed and plastic trays loaded with gravel and egg-filled Whitlock-Vibert boxes were submerged to the riverbed in pools located at the head of riffles. Survival of eggs

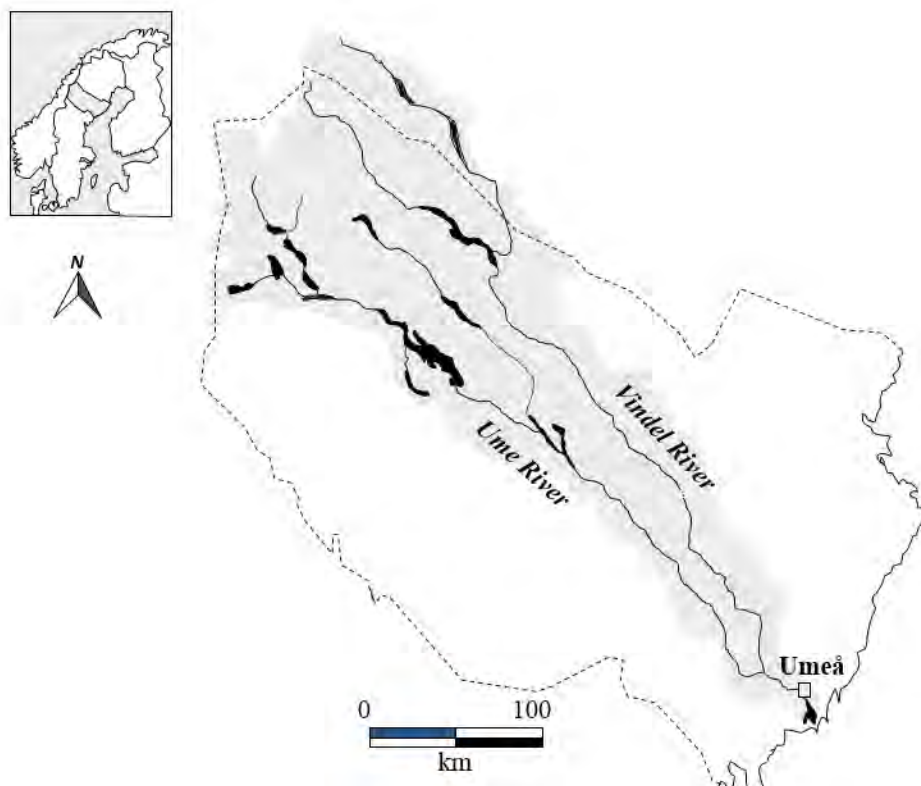


Figure 1. Map showing the location of the Ume and Vindel rivers.

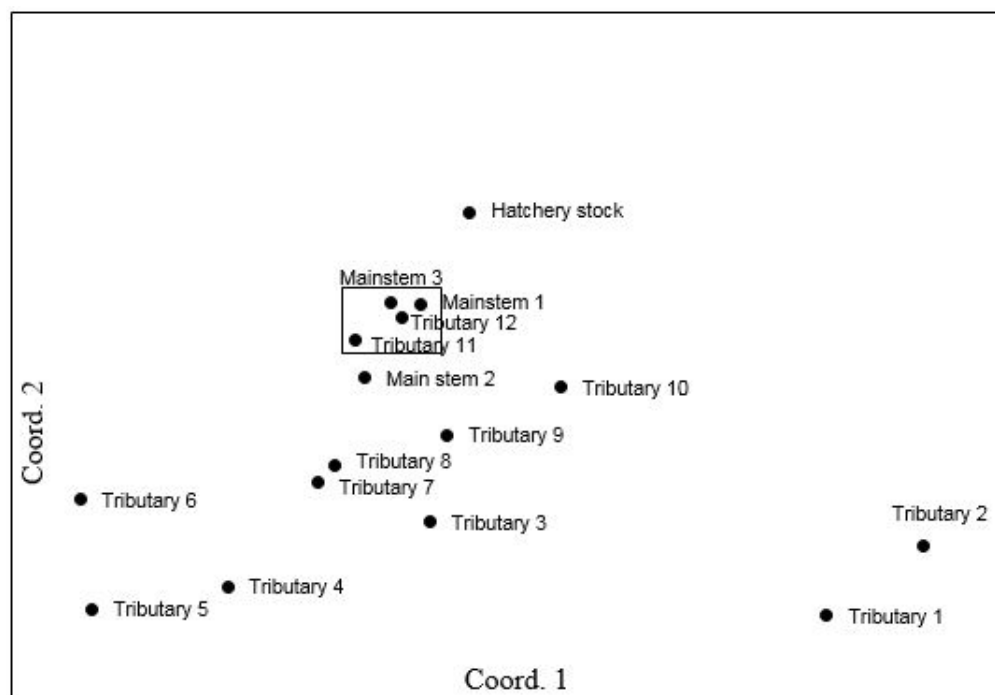


Figure 2. Principal coordinates (PCoA) of Single Nucleotide Polymorphism variation of trout from 15 sites along the Vindel River and its tributaries. The rectangle indicate sites in the main stem and tributaries where subpopulations are closely related and not genetically differentiated from the hatchery stock. The hatchery stock consists of ascending anadromous Brown Trout.

was estimated using electrofishing in August in riffles located in the vicinity of stocking sites. Dispersal of fry was evaluated by tissue samples from trout caught during electrofishing which were matched genetically (SNP variation) to stocked eggs.

The SNP panel was developed by RAD sequencing. The assays were tested and validated on the original samples sent in for sequencing and known kinship samples, including replicates, to ensure markers reliably genotype according to Mendelian expectations. The error rate is <0.001 .

Results and Discussion

So far, fifteen subpopulations have been genetically distinguished (Figure 2). The hatchery stock, which consists of ascending anadromous individuals, has also been analyzed. Four subpopulations are regarded closely related to the

hatchery stock and are therefore not likely to be disturbed by stocking. Survival rates of stocked eggs appeared to be good. Egg mortality was low and high densities of juvenile trout were found at all sites where stocking had been conducted. Studies of dispersal show that most individuals remain within 200 m from the stocking point during the first 2 years. This pattern of dispersal corresponds well to previous studies of wild trout and salmon.

These early results indicate that stocking of eyed eggs can be a complementary tool to trout restoration in the Vindel River. However, further mapping of genetic variability must be carefully conducted before large scale stocking programs can be started without putting valuable native genes to risk. Furthermore, detailed studies on the survival and dispersal of stocked individuals must be conducted and compared to wild individuals in order to predict to what extent stocking may contribute to trout restoration.



A Long-Term Watershed-Scale Partnership to Restore Bull Trout in Sun Creek

Dave Hering

Crater Lake National Park

Abstract—For 25 years, Crater Lake National Park has used exclusion barriers, electrofishing, and piscicide to remove nonnative fish that competed with ESA-listed Bull Trout *Salvelinus confluentus* in Sun Creek. Since we presented early results of this work at the Wild Trout VII symposium 17 years ago, the project has grown to include multiple agency and nongovernmental partners working across administrative boundaries to achieve watershed-scale restoration goals. Nonnative trout have been eradicated from 23.5 km of Sun Creek on national park and State forest land, and partners have worked with private landowners to restore stream habitat, transfer water rights, screen diversions, and improve irrigation efficiency. The Bull Trout population within the national park has expanded to renovated habitat downstream and is now connected to the surrounding stream network through a restored migratory corridor. Soon, this recovering population may serve as a source for active reintroduction to other historically-occupied watersheds in the upper Klamath basin. The growth and effectiveness of this project required sustained commitment to restoration, which has been reinforced by the persistent conservation mission of the National Park Service. The project exemplifies how partnerships can build on that mission to restore native trout on a landscape scale.



Utah's North Slope Uinta Colorado River Cutthroat Trout Brood: Lessons and Progress After 15 Years

Bryan Engelbert¹, Garn Birchell²

¹Regional Aquatics Biologist, Utah Division of Wildlife Resources, 318 N. Vernal Ave, Vernal, UT 84078, 435-219-6525, bengelbert@utah.gov,

²Assistant Aquatics Manager, Utah Division of Wildlife Resources, 318 N. Vernal Ave, Vernal, UT 84078, 435-823-0963, garnbirchell@utah.gov

Abstract—Northeast Utah's sole native salmonid is the Colorado River Cutthroat Trout (CRCT) *Oncorhynchus clarkii pleuriticus*. Several factors in the modern era have negatively impacted this alluring fish that include nonnative species introductions, habitat loss, disease, and drought. State and federal stakeholders proactively formed the Conservation Agreement and Strategy for Colorado River Cutthroat Trout to combat this decline. A portion of this Strategy formed the Geographical Management Units (GMU's) as a basis to conserve the localized, unique genetic traits in individual CRCT populations. One of the underlying strategies for species recovery is to create wild broodstocks in each of these GMU's to secure fish sources, create sportfishing opportunities, and create an ability to supplement or repopulate conservation populations. We examine our current status and delve into lessons learned with efforts to create a wild CRCT broodstock on the North Slope of the Uinta Mountains in Northeast Utah. In the 15 years of developing this brood, we have faced issues of faulty genetics testing, impure genetics, disease, small fish size, low fecundity, small fish population size, remote harvesting locations, low re-capture efficiency, low egg survival, and time and money hurdles. We are currently working through various means to minimize or combat these issues and preserve our native fish heritage.

Introduction

The Colorado River Cutthroat trout (CRCT) *Oncorhynchus clarkii pleuriticus* currently occupies approximately 14% of its historic range based on the most current available data gathered by state fish and wildlife and federal land management agencies (CRCT Conservation Team 2006). Reasons for this decline include habitat loss, disease, and drought, with nonnative fish species competitive suppression and hybridization arguably having the greatest impact (Young et al. 1996; CRCT Conservation Team 2006). In 1999, several groups petitioned the U.S. Fish and Wildlife Service (FWS) to list the CRCT under the Endangered Species Act (ESA). The FWS found listing to be unwarranted in 2004 and 2007, citing current species' status and conservation agreements (Lentsch and Converse 1997) in place at that time (U.S. Fish and Wildlife Service 2004; 2007).

The states of Colorado, Wyoming, and Utah, along with the FWS, U.S. Forest Service, National Park Service, Bureau of Land Management, Ute Indian Tribe, and Trout Unlimited entered into the CRCT

Conservation Agreement and Strategy (2001) to preemptively act to preserve and perpetuate the CRCT (CRCT Conservation Team 2006; CRCT Coordination Team 2006). The Conservation Agreement and Strategy exist in two separate documents that are typically updated every 5 years; both documents are currently under review in 2017. The Conservation Agreement and Strategy sets forth objectives, strategies, goals, and responsibilities of the signatories in order to establish a methodical approach toward the species' restoration (CRCT Conservation Team 2006; CRCT Coordination Team 2006). The CRCT Conservation Strategy outlines 11 strategies for signatories to undertake. We will focus on Strategy 4: Maintain sources of genetically pure Colorado River Cutthroat Trout. This Strategy requires creation of at least one brood per unique area or GMU to ensure localized genetics and fitness be represented within populations. The theme of this manuscript is to create and sustain a broodstock for the North Slope of the Uinta Mountains (Figure 1). We discuss obstacles, progress, and lessons learned through the 15 years of creating this broodstock.



Figure 1. North Slope Uinta Mountains of Utah (highlighted) within the Colorado River drainage.

Methods and Results

Any wild fish populations considered for wild brood purposes in Utah must pass Utah Fish Health Board policies. Our policy dictates that three clean disease certifications of at least 60 fish per certification be undertaken in a 3-year time period (Rule R58-17. Aquaculture and Aquatic Animal Health [<https://rules.utah.gov/publicat/code/r058/r058-017.html>]). More recent policy changes have garnered flexibility by requiring at least 4 months between each certification, thereby speeding up the process of certifying a water. We transfer or spawn fish if certified as a clean CRCT population. Unfortunately, whirling disease has spread to numerous waters in our region and has infested numerous conservation populations of CRCT in Northeast Utah. Fish health policy allows for gamete collection (no transfers) in cases where we need to spawn wild CRCT where whirling disease is prevalent.

The DWR currently owns water and facility rights to Sheep Creek Lake. We operate this facility under Special Use Permit from the U.S. Forest Service,

Ashley National Forest. The DWR constructed Sheep Creek Lake for Cutthroat Trout conservation in 1959. The DWR rededicated this facility to CRCT conservation in 1999 for the North and South Slope Uinta CRCT broodstocks. Sheep Creek Lake is an ideal spot for wild broodstock since the DWR can directly manipulate water discharge variables, the inlet canal already possessed a working fish trap, and the lake is productive enough to yield large adult individuals.

The DWR initially attempted to establish broodstock in Sheep Creek Lake via fish transfers to the lake from stream CRCT populations. The West Fork Duchesne and the West Fork Smiths Fork were chosen as ideal candidates to represent the South and North Slope Uinta broods, respectively. The DWR initiated the disease certification for West Fork Smiths Fork in 2001. A clean health report in 2003 meant that the DWR could transfer CRCT from this source. Table 1 details the number of individual fish transferred from the West Fork Smiths Fork from 2003 to 2006, subsequent success of fish recapture in the fish trap

at Sheep Creek Lake, and the number of eggs or fingerlings that resulted each year from 2003 to 2008.

Genetics testing completed by the University of Montana yielded that the West Fork Smiths Fork CRCT population was 99.6% pure (DWR internal documents). The DWR decided to retest numerous important CRCT populations with the advent of nuclear DNA testing. The West Fork Smiths Fork was re-tested in 2006; this study determined that this population was 85% pure, with 14% Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* introgression (Evans and Shiozawa 2007). Due to the significance of the find in combination with the resources already spent on procuring a brood from the West Fork Smiths Fork, the DWR re-tested this population in 2007. The study found similar results of 89% CRCT with 11% Yellowstone Cutthroat Trout introgression (Evans and Shiozawa 2008). After a review, it was determined that the 1999 genetics testing only sampled for Rainbow Trout *Oncorhynchus mykiss* hybridization. Upon receiving the results of the second testing, all North Slope fish within the hatchery system and Sheep Creek Lake were disposed after failing to meet conservation population criteria (>90% purity).

The DWR investigated several other North Slope Uinta CRCT sources in 2007-2008. These included the Burnt Fork, the Little West Fork Blacks Fork, Elk Creek, and North Fork Sheep Creek. The latter three tested 100% pure, with the Burnt Fork slightly less pure at 98%. However, each stream represented differing challenges to procuring a broodstock. These challenges included small fish, small population size, inaccessibility, and whirling disease. Disease certification started in 2009 in Little West Fork Blacks Fork and Elk Creek to pursue fish transfers to Sheep Creek Lake. These were the most accessible streams of our limited list; however, they also contained the smallest populations. The DWR transferred approximately 100 fish cumulatively per year from Little West Fork Blacks Fork and Elk Creek to Sheep Creek Lake in 2013-2015. As of date of publication, none of these fish have returned to the trap at Sheep Creek Lake; however, fish in 2014-2015 were spawned prior to transfer, which yielded limited gametes. The DWR first pursued disease certification in the North Fork Sheep Creek in 2013. This stream has whirling disease and thus all management actions are limited to streamside fish spawning and egg collection. The

Table 1. Number of fish transferred to Sheep Creek Lake and progeny produced from West Fork Smiths Fork Cutthroat Trout. Fish reared at the Fisheries Experimental Station State Fish Hatchery, Logan Utah. Fish were destroyed at the hatchery in 2008 due to nonative species hybridization.

Year	Number of Fish Transferred	Number of Fish Captured in Trap	Fingerlings Produced from Trap
2003	155	0	0
2004	106	15	0
2005	148	9	150
2006	180	9	211
2007	0	12	100
2008	0	17	*10,881 eggs were dumped post-hatch

Table 2. North Slope Uinta Colorado River Cutthroat Trout fish production data 2014-present.
 Fingerlings from 2014 and 2015 were reared at Whiterocks State Fish Hatchery, Whiterocks, Utah, and subsequently stocked into wild brood source locations. Fingerlings from 2016 and 2017 spawns are currently at the Fisheries Experimental Station State Fish Hatchery, Logan, Utah.

Year	Source	Number of Females Spawned	Eggs	Fingerlings	Hatch Success
2014	LWF Blacks Fk Elk Creek	28	1,179	270	22.9%
2015	LWF Blacks Fk Elk Creek NF Sheep Ck	39	3,519	2,000	56.8%
2016	Lost Lake	28	6,144	2,500	40.7%
2017	LWF Blacks Fk Lost Lake NF Sheep Ck	46	14,758	5,827	39%

first wild fish spawn took place in 2015, with resulting fingerlings stocked into Sheep Creek Lake in fall 2016. In 2015, the DWR was able to issue fish health policy variances on a case-by-case basis for conservation species management actions. Following variance protocol of not transferring fish outside of the drainage they occurred in, we transferred 555 CRCT from North Fork Sheep Creek into Lost Lake. Lost Lake is a montane lake approximately three miles away from the North Fork Sheep Creek. Additionally, we also stocked 180 CRCT from the 2014 fish spawns into Lost Lake. We chose Lost Lake as an additional broodstock lake since it passed the fish health policy variance, and it was recently vacated by piscicide treatment. We believed that within the lake environment, the stocked CRCT would grow larger and thus become more fecund than what they could achieve in a stream environment. Our observations significantly agree with this finding. Table 2 details specific results for number of females spawned per year from which individual sources, eggs collected from those efforts, and resulting fingerlings hatched.

Conclusions and Implications

Many factors need to be considered before initiating a fish brood for conservation purposes. In

this broodstock's history, several factors were missed, unaccounted for at inception, or simply needed years of intensive physical labor for it to be successful. These factors include genetic purity and diversity, disease concerns or issues within a fish population, small fish size and low fecundity, small fish population size, location and proximity to accessible areas, recapture efficiency of transferred or released individuals, ability of stocked individuals to grow, egg survival during wild egg take operations, and time and money expenses that an agency will need to provide for successful operations. Inaccurate assessment of genetic purity cost the DWR 7 years of productivity while working with West Fork Smiths Fork fish. The DWR has regularly had to work with whirling disease positive populations in order to achieve our goals; we deemed streamside fish spawning operations to be worth the effort for creating a conservation population. We have broached problems of small fish size and low fecundity by introducing fish into a lake environment where we believe we can have a high recapture efficiency while the fish are able to get larger. Meanwhile, we have simply used more effort to capture more individuals in order to produce enough gametes. Through this broodstock creation, the DWR has dealt with small fish population size by

carefully removing enough individuals or gametes to have a chance at being effective while not removing enough to cause a bottleneck or population crash. By incorporating individuals or gametes from multiple population sources we are also increasing genetic diversity of the brood. While the two small streams, Little West Fork Blacks Fork and Elk Creek are relatively accessible, the DWR has spent significant amounts of time and money to pursue the gamete collection via hiking and horseback on North Fork Sheep Creek. The North Fork collection sites are approximately 3 mi from a trailhead, but has paid dividends in terms of fish production that would not otherwise be accomplished. We have not regularly monitored or tested recapture efficiency. We have relied on the fish trap to recapture individuals at Sheep Creek Lake. However, we feel it is prudent advice to try other more active sampling methods in cases where recapture rates are extremely low.

Other issues encountered in forming this brood include a mismatch between male and female spawn times and considering options to increase extending egg takes to ensure all adult fish be successfully spawned. Beginning with the North Fork Sheep Creek streamside egg take in 2015, we noted a substantial mismatch between male and female spawn times. Males were peaking at approximately the third week of June, while relatively few females were ready at that time. Females peaked in the second week of July and the majority of males were spent at that time. We noted similar results during the Lost Lake spawn in 2016. In June 2017, we conducted a small study to use cryopreserved milt to fertilize females approximately 2 weeks after we initially collected the milt. We did not have any egg survival. We will continue to explore this methodology for future practical use in order to artificially allow more wild fish spawns when there is a mismatch in spawn timing between genders. Obviously in cases where adult fish are difficult to collect in these conservation populations, biologists will want to collect as many viable gametes as possible.

In 2015, the DWR set in motion the idea of rededicating a smaller hatchery facility for wild trout

broodstock rearing. The results of our 2016 and 2017 egg takes are currently in our hatchery system. In cases where wild egg takes and locations become too taxing or expensive but an agency has the available hatchery space for rearing adult fish, it may be prudent to explore the feasibility of a captive broodstock to meet fish quota demands rather than collecting from wild brood sources. This option may be discontinued once the brood population is high enough to the point to have highly successful recaptures in the wild. However, caution should be used when using captive broodstocks in order to avoid domestication. We recommend the captive broodstock be replaced with progeny of wild fish in order to avoid this problem.

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Describing Spatial Variability in Streams of the Coeur d'Alene Lake Basin, Idaho, Using Strontium Isotopes

J. W. Heckel IV¹, C. J. Watkins², A. M. Dux³, M. C. Quist⁴, S. A. Carleton⁵

¹M. S. Student, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Drive MS1141, Moscow, ID 83844, 307-413-8759, heck5910@vandals.uidaho.edu.

²Idaho Department of Fish and Game, Coeur d'Alene, Idaho, 2885 W. Kathleen Ave, Coeur d'Alene, ID 83815, 208-769-1414, carson.watkins@idfg.idaho.gov.

³Idaho Department of Fish and Game, Coeur d'Alene, Idaho, 2885 W. Kathleen Ave, Coeur d'Alene, ID 83815, 208-769-1414, andy.dux@idfg.idaho.gov.

⁴United States Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Drive MS1141, Moscow, ID 83844, 208-885-4064, mcquist@uidaho.edu.

⁵Chief, Division of Migratory Birds, U. S. Fish and Wildlife Service, Region 2, 500 Gold Avenue, SW, Albuquerque, NM 87102, 505-248-6639, scott_carleton@fws.gov.

Abstract—Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) are widely distributed throughout the Coeur d'Alene Lake basin and exhibit resident, fluvial, and adfluvial life history strategies. Although the basin supports all of these life history strategies, sources of recruitment and natal origins for adfluvial WCT populations throughout the Coeur d'Alene Lake basin are poorly understood. The objective of this study was to assess spatial variability in strontium (Sr) stable isotopes throughout the Coeur d'Alene Lake basin by referencing ⁸⁷Sr/⁸⁶Sr stable isotope ratios across otolith growth axes using laser ablation multicollector inductively coupled plasma mass spectrometry (LA-MC-ICPMS). Changes in ⁸⁷Sr/⁸⁶Sr ratios across the growth axis of an otolith provide insight into life histories of sampled fish. Sagittal otoliths were extracted from 310 fish sampled from 41 different locations throughout the basin and analyzed for ⁸⁷Sr/⁸⁶Sr stable isotope ratios. Data analyzed from sagittal otoliths suggest that there is variability in ⁸⁷Sr/⁸⁶Sr isotope ratios within and among watersheds in the Coeur d'Alene Lake basin. Results from this study provide a foundational understanding of WCT distribution, population structure, and sources of recruitment throughout the Coeur d'Alene Lake basin, which will help to enhance management of WCT in the basin at large.

Introduction

In the panhandle region of Idaho, Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) are native in the Spokane River upstream of Spokane Falls into Coeur d'Alene Lake and its tributaries (Behnke 1992). Populations of WCT in the Coeur d'Alene Lake basin appear to be thriving, but little is known about the population dynamics, distribution, and life history characteristics of WCT in the system. Some research has been conducted in the Coeur d'Alene Lake basin to better understand WCT abundance and distribution (Jeppson 1960; Averett 1962; Lukens 1978; Thurow and Bjornn 1978; Rieman and Apperson 1989; Wells et al. 2004; Prometrix 2005; Firehammer 2012). However, few lotic systems supporting WCT have been thoroughly investigated; most comprehensive studies are decades old and there is a knowledge gap regarding life history

structure in the Coeur d'Alene Lake basin. Research on WCT in the Coeur d'Alene Lake basin indicates that movement among lotic systems and between lotic-lentic environments contribute to a broad distribution and to the persistence of WCT populations. Where these population linkages exist and the extent to which adfluvial WCT use the Coeur d'Alene Lake basin at-large is essentially unknown.

The Coeur d'Alene Lake basin is located in the panhandle of northern Idaho and drains an area of about 9,946 km². Approximately 27 tributaries flow into Coeur d'Alene Lake; the two principle tributaries are the Coeur d'Alene River and the St. Joe River (Figure 1). Linking fish movement patterns to landscape and aquatic habitat has long been an important challenge in fisheries research and management (Schlosser 1995; Fausch et al. 2002; Wells et al. 2003). Otolith microchemistry

is a useful tool for inferring migration history, life history variation, maternal origins, stock assessment, and natal origins of freshwater and marine fishes (Campana 1999; Volk et al. 2000; Barnett-Johnson et al. 2008; Muhlfeld et al. 2011; Paracheil et al. 2014). Isostructural to calcium (Ca), strontium (Sr) replaces Ca in geological and biological structures. Rock weathering and geologic processes dissolve Sr and other ions in water. Ions from ambient water are transferred to a fish's blood plasma. Therefore, relatively high concentrations of Sr—that reflect the geology of a drainage—are incorporated into the otoliths of fish as they form. $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios are consequently unique geochemical signatures derived from the water where fish live. Chemical signatures from otoliths can then be used to assess the variability among and within watersheds, to evaluate maternal origins, natal origins, migration history, and assess life history structure of a population (Wells et al. 2003; Pangle et al. 2010; Muhlfeld et al. 2011; Paracheil et al. 2014; Chase et al. 2015).

Methods

Sampling Design

We used a stratified sampling design to characterize the Sr isotopic variability of the Coeur d'Alene Lake basin; 50 locations were selected to collect age-0 and/or age-1 WCT and 50-ml water samples. Sampling reaches were randomly chosen in the field from preselected tributaries. Kokanee *Oncorhynchus nerka* were used as a surrogate for WCT to obtain an $^{87}\text{Sr}/^{86}\text{Sr}$ signature from Coeur d'Alene Lake.

Sampling Protocol

A Smith-Root LR-24 battery-powered backpack electrofishing unit was standardized to 100 watts of pulsed direct current (PDC) power output according to stream conductivity ($\mu\text{S}/\text{cm}$; Bertrand et al. 2006). All fishes were collected and identified in the field. Electrofishing continued until 10 age-0 or age-1 WCT were collected, or until 3,600 s of electrofishing effort

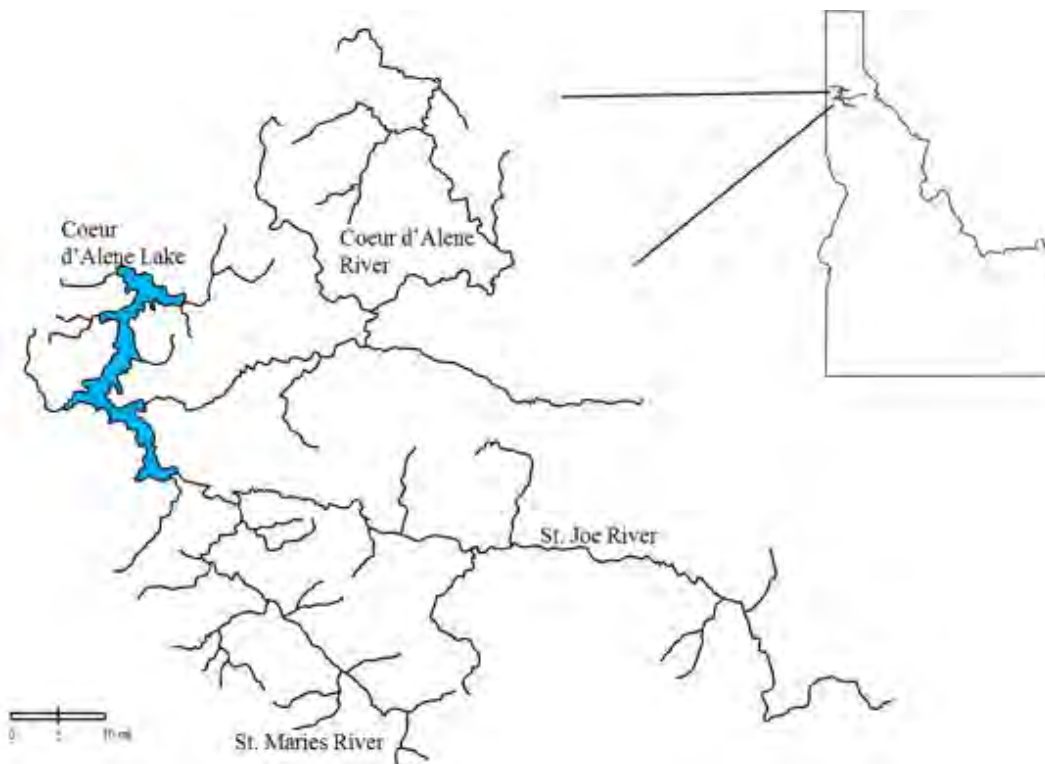


Figure 1. Coeur d'Alene Lake and major river systems: Coeur d'Alene River, St. Joe River, and St. Maries River with tributaries that were sampled during the study.

were completed (McGrath et al. 2008). Cutthroat Trout were euthanized using a lethal dose of buffered Tricane-222, then stored in Whirl-Paks and kept on ice until they were frozen in the lab. At sampling locations where WCT were not captured or present, other age-0 salmonids (e.g. Brook Trout *Salvelinus fontinalis*) served as surrogate species. When no salmonids were captured, the stream signature was based solely from the water sample taken from that sampling reach (Kennedy et al. 2000). To obtain a signature from Coeur d'Alene Lake, four locations in the lake were sampled for kokanee using hook-and-line sampling and water samples were concurrently taken. Total length was recorded and kokanee were euthanized in the field and sagittal otoliths removed. Sagittal otoliths were removed, placed dry in 1.5-ml polypropylene vials, and stored in coin envelopes with date, length, location, and unique identification number recorded on the envelope.

Water Sampling

Vials (50 ml polypropylene), lids, and syringes (10 ml polypropylene) used for water sampling were washed using a 6N hydrochloric acid bath for 2 h, followed by three rinses with ultrapure water (18.20 MΩ * cm), then washed in a 1% trace metal grade nitric acid bath for 24 h, rinsed three times with ultrapure water, air dried, then stored in sterile Whirl-Paks. Water was filtered through 25-mm diameter, 2-μm syringe filters. Water samples were collected at baseflow at the downstream end of a sampling reach to capture the Sr variability of upstream sources (Hegg et al. 2013). Water samples will be analyzed for $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios and 27 different trace elements using inductively coupled plasma mass spectrometry (ICPMS) with Nu Plasma HR (Nu032) multiple-collection, high-resolution, double-focusing plasma mass spectrometer system at the University of California-Davis Interdisciplinary Center for Plasma Mass Spectrometry.

Laboratory Protocol

Total length was recorded and sagittal otoliths from WCT were removed in the lab. Otoliths were placed in dry 1.5-ml polypropylene vials, and stored in coin envelopes. One otolith per fish was mounted with the sulcus acusticus side up on a microscope slide using Crystalbond 509-3. Mounted otoliths were

then wet-sanded using ultrapure water on a Buehler MetaServ 250 grinder-polisher with 800-1,200 grit sandpaper. Otoliths were sanded until incremental growth from the otolith core (primordium) to the edge was exposed (Thorrold et al. 1998; Hobbs et al. 2010; Chase et al. 2015). A compound microscope was used in conjunction while sanding to assess when daily incremental growth was visible at 400× to 1000× magnification. Otoliths were then mounted onto petrographic slides with Crystalbond 509-3 for laser ablation multicollector inductively coupled plasma mass spectrometry (LA-MC-ICPMS). Otoliths were analyzed using LA-MC-ICPMS for $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios with transect line scans (scan speed 5 μm/s, depth 5 μm, line spot size 40 μm) ablated across the dorsoventral plane of the otolith. Line scans ablated from dorsal edge to ventral edge and passed through the core. This method captures changes in $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios across the otolith growth axis. Sr values from the most recent otolith growth increments (20 μm to 100 μm from edge; adjusted for size of the otolith) are termed the edge values. Edge values were averaged for each otolith and the mean $^{87}\text{Sr}/^{86}\text{Sr}$ value was classified as the stream signature for that fish. The stream signature for each stream was evaluated from the mean of all otoliths from that particular stream. In female fishes, ions are transferred from blood plasma into eggs that are developing in ovaries, and are consequently inherited into the fluid of a yolk sac (Campana 1999; Campana and Thorrold 2001; Barnett-Johnson et al. 2008). Fluid in the yolk sac becomes part of the endolymphatic fluid as a fish develops into larvae. Endolymphatic fluid encases otoliths: as such, elements in endolymphatic fluid are incorporated into the crystalline aragonite matrix of otoliths. This translates to an $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratio value from the otolith core (primordium), or pre-hatch region, that is linked to the ambient water where the mother lived prior to spawning (Bacon et al. 2004). The isotope value at the core was considered the maternal signature.

Results and Discussion

Fifty ($n = 50$) different locations were sampled in 2016 throughout the Coeur d'Alene Lake basin. Fish were collected and used for microchemistry analysis from $n = 41$ locations. Fish collected and used for microchemistry analysis included Brook Trout ($n = 9$),

kokanee ($n = 27$), and WCT ($n = 274$). Sagittal otoliths were analyzed for $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios on $n = 310$ fish. To obtain a signature for Coeur d'Alene Lake kokanee sagittal otoliths were analyzed. Juvenile WCT were present at $n = 36$ locations and sagittal otoliths were analyzed for $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios. Water sample analysis for $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios are forthcoming. Results of laser ablation analysis for $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios on sagittal otoliths are summarized in Table 1.

Stream and lake signatures were inferred by taking the mean of $^{87}\text{Sr}/^{86}\text{Sr}$ from the most recent otolith growth, which ranged from 50 μm to 200 μm from the edge to the core. Maternal signatures were deduced from the mean of $^{87}\text{Sr}/^{86}\text{Sr}$ at the core (primordium), or pre-hatch, region of the otolith. Based on these data, it is possible to discriminate within and among watersheds throughout the Coeur d'Alene Lake basin using $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios. One limitation of these results is a lack of comparison between water samples (solution based sample) and sagittal otoliths (hard part sample). Kokanee from this study were sexually mature fish that had only lived in Coeur d'Alene Lake. Therefore, kokanee otoliths were hypothesized to display variability within Coeur d'Alene Lake; however, these data indicate that there is little variability within Coeur d'Alene Lake. Contrarily, kokanee may have low site fidelity and represent the lake at-large rather than the locations of the lake where they were caught. Age-0 or age-1 WCT were selected for laser ablation analysis so that recent otolith growth and the primordium region comprised the majority of the structure.

We established that WCT are widely distributed throughout the Coeur d'Alene Lake basin, and production is occurring in 36 of the 46 stream location that we sampled (based on presence of age-0 or age-1 WCT). Analyzing the primordium region of otoliths resulted in our finding that resident, fluvial, and adfluvial populations exist throughout the Coeur d'Alene Lake basin. Most importantly, from our samples we concluded that Beauty Creek, Carlin Creek (CDAL), Latour Creek, Marble Creek, North Fork St. Joe River, Shoshone Creek, West Fork Merry Creek, and Wolf Lodge Creek all have influences from adfluvial WCT. All other streams we sampled showed evidence of both fluvial and resident WCT populations based on maternal signatures.

We found that heterogeneity exists among and within basins caused by variances in underlying geology. Our findings are similar to Wells et al. (2003) who quantified molar ratios of magnesium, strontium, barium, and manganese to calcium derived from WCT sagittal otoliths in streams of the Coeur d'Alene River basin. Wells et al. (2003) also found a linear relationship between water chemistry and otolith chemistry, suggesting that otolith chemistries can represent the water chemistries of a given stream. Furthermore, they found that elemental ratios varied significantly among streams such that they could reclassify individual fish to streams from which they were collected with 100% accuracy (Wells et al. 2013).

Aside from analyzing Sr isotope ratios and not elements to calcium, a new finding from our research is the Coeur d'Alene Lake $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratio derived from kokanee otoliths (~ 0.7350). A lake signature that is unique from all other sampled streams provides us with evidence that it is possible to determine adfluvial maternal signatures. In addition to a unique lake signature, we detected heterogeneity in $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios among most streams that were sampled throughout the Coeur d'Alene Lake basin [e.g. within the St. Maries River watershed Thorn Creek and Alder Creek (t_{10} , $\alpha = 0.05$, $P = 8.144\text{e}^{-13}$)], which suggests that stream origins could be extrapolated from adult WCT sagittal otoliths and identified at the drainage scale. Our research expands on the Wells et al. (2003) findings because we analyzed exclusively for Sr isotope ratios, we researched at a larger spatial scale, and we used a larger sample size of both fish and sampling locations.

Our work indicates that we can describe and differentiate spatial variability within and among watersheds in the Coeur d'Alene Lake basin using $^{87}\text{Sr}/^{86}\text{Sr}$ ratios derived from WCT sagittal otoliths. These data indicate that juvenile fish can be linked to the stream where they were collected, but the stream origin of the maternal signature may not be easily identifiable if the mother is a fluvial migrant because some streams have statistically similar signatures. Data from this project can be used to address movement patterns of WCT in the Coeur d'Alene Lake basin. Additionally, the heterogeneity among and within watersheds in this system can be used to better understand the life histories of WCT, their natal origins, and maternal origins. Estimates of relative contribution that particular streams have on various

Table 1. Summary of LA-MC-ICPMS analyses of $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratios from sagittal otoliths extracted from WCT and kokanee. $^{87}\text{Sr}/^{86}\text{Sr}$ isotope ratio signatures for stream and lake locations sampled throughout the Coeur d'Alene Lake basin in 2016.

Location	Stream signature $^{87}\text{Sr}/^{86}\text{Sr}$	Standard error	Sample size (n)
St. Maries River basin			117
Canyon Creek	0.7086	0.0000	12
Thorn Creek	0.7094	0.0000	10
Carlin Creek (SMR)	0.7123	0.0000	15
SF Santa Creek	0.7138	0.0000	10
WF St. Maries River	0.7139	0.0004	13
Alder Creek	0.7147	0.0001	10
Cats Pur Creek	0.7197	0.0003	7
Hume Creek	0.7210	0.0003	11
EF Charlie Creek	0.7295	0.0003	2
WF Emerald Creek	0.7307	0.0002	5
Childs Creek	0.7344	0.0005	10
WF Merry Creek	0.7347	0.0002	8
Renfro Creek	0.7507	0.0015	4
St. Joe River basin			63
Marble Creek	0.7280	0.0002	10
Big Creek	0.7333	0.0002	5
Hugus Creek	0.7383	0.0002	5
NF St. Joe River	0.7430	0.0005	7
Bluff Creek	0.7538	0.0026	5
St. Joe River headwaters	0.7582	0.0020	11
St Joe River mid	0.7607	0.0006	4
St. Joe River upper	0.7607	0.0008	7
St Joe River lower	0.7617	0.0003	4
Gold Creek	0.7651	0.0012	5
Coeur d'Alene River basin			30
Shoshone Creek	0.7192	0.0001	7
Jordan Creek	0.7198	0.0001	5
Teepee Creek	0.7211	0.0001	5
Little NF Coeur d'Alene River	0.7234	0.0002	3
Independence Creek	0.7242	0.0010	4
Latour Creek	0.7253	0.0001	6
Coeur d'Alene Lake			100
SF Mica Creek	0.7134	0.0000	9
Benewah Creek	0.7147	0.0014	8
NF Mica Creek	0.7164	0.0000	5
Lake Creek	0.7167	0.0006	13
Cougar Creek	0.7174	0.0000	10
Wolf Lodge Creek	0.7206	0.0001	5
Carlin Creek (CDAL)	0.7273	0.0001	12
Beauty Creek	0.7274	0.0001	11
Coeur d'Alene Lake section 1	0.7352	0.0002	10
Coeur d'Alene Lake section 4	0.7353	0.0004	4
Coeur d'Alene Lake section 2	0.7356	0.0002	8
Coeur d'Alene Lake section 3	0.7357	0.0002	5
Total			310

life history strategies contributes to our understanding of which streams are most important for adfluvial, fluvial, or resident WCT. Taking this understanding one step further allows us to make more informed management decisions regarding the WCT population in the Coeur d'Alene Lake basin at-large. Therefore, we can now begin to understand which streams have more relative production and recruitment to the Coeur d'Alene Lake WCT population.

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Session 6

Nonnative Fishes and Tools for Native Trout Management





Changes in Species Composition from Native Yellowstone Cutthroat Trout to Nonnative Species: Implications for Management and Conservation

Robert Al-Chokhachy

US Geological Survey Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, MT 59715

Extended Abstract

Nonnative species are increasingly considered one of the largest threats to native trout populations species (Muhlfeld *et al.*, 2009; Cucherousset and Olden, 2011; Roberts *et al.*, 2017). Our understanding of how the distribution and abundance of nonnative species is changing through time, however, is often limited by available data spanning relevant spatial and temporal scales. Quantifying changes in nonnative populations and the factors driving those changes is an important component of Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* conservation as 70% of all extant populations are either slightly hybridized or sympatric with nonnative species.

Here, we use datasets spanning multiple decades from the Shields River, Montana (1974 -2016) and the Teton River, Idaho (2015-2015) basins to consider temporal changes in the distribution and abundance of native Yellowstone Cutthroat Trout and nonnative Brook Trout *Salvelinus fontinalis*, Rainbow Trout *O. mykiss*, and Brown Trout *Salmo trutta*. Our analyses included data from sites distributed throughout each basin as well as long term trend data from main-stem sites.

Our results clearly demonstrate three general patterns. First, despite finite evidence of extirpations of Yellowstone Cutthroat Trout and colonizations of nonnative trout, the majority of sites demonstrated no shifts in species distribution patterns over the past 15-20 years (e.g., Figure 1); changes in distribution were more evident when considering temporal periods of 20-40 years, where data were available. This result highlights the challenges of using distribution data in status assessments. Second, despite little shift in distribution, we found a large number of sites with increases in the proportion of nonnative trout, particularly over the past decade (Figure 2). The changes in proportion of nonnatives varied considerably across basins and sites, however. Interestingly, the observed shifts, however do not conform to our current understanding of how temperature, among other factors, is driving changes in communities; in particular, we observed nonnatives

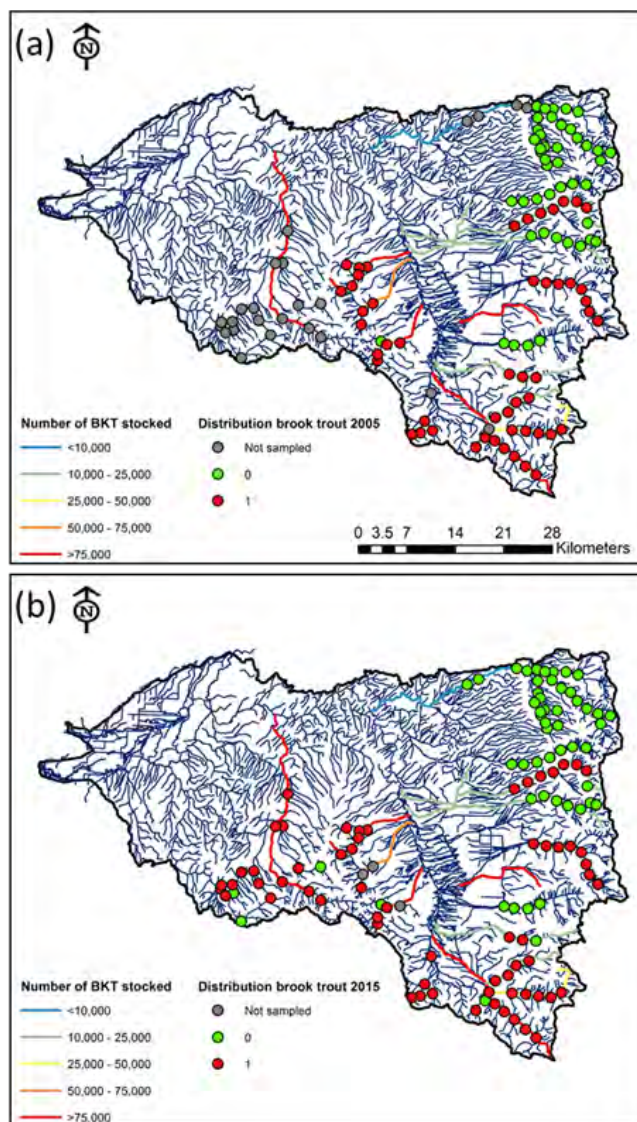


Figure 1. The number of Brook Trout stocked (cumulative) and the presence (red) and absence (green) of nonnative Brook Trout in the Teton River basin, Idaho, in 2005 (a) and 2015 (b).

dominating species composition at sites with cold summer temperatures ($<8^{\circ}\text{C}$). Finally, we observed considerable shifts in the population structure of Yellowstone Cutthroat Trout at long term monitoring

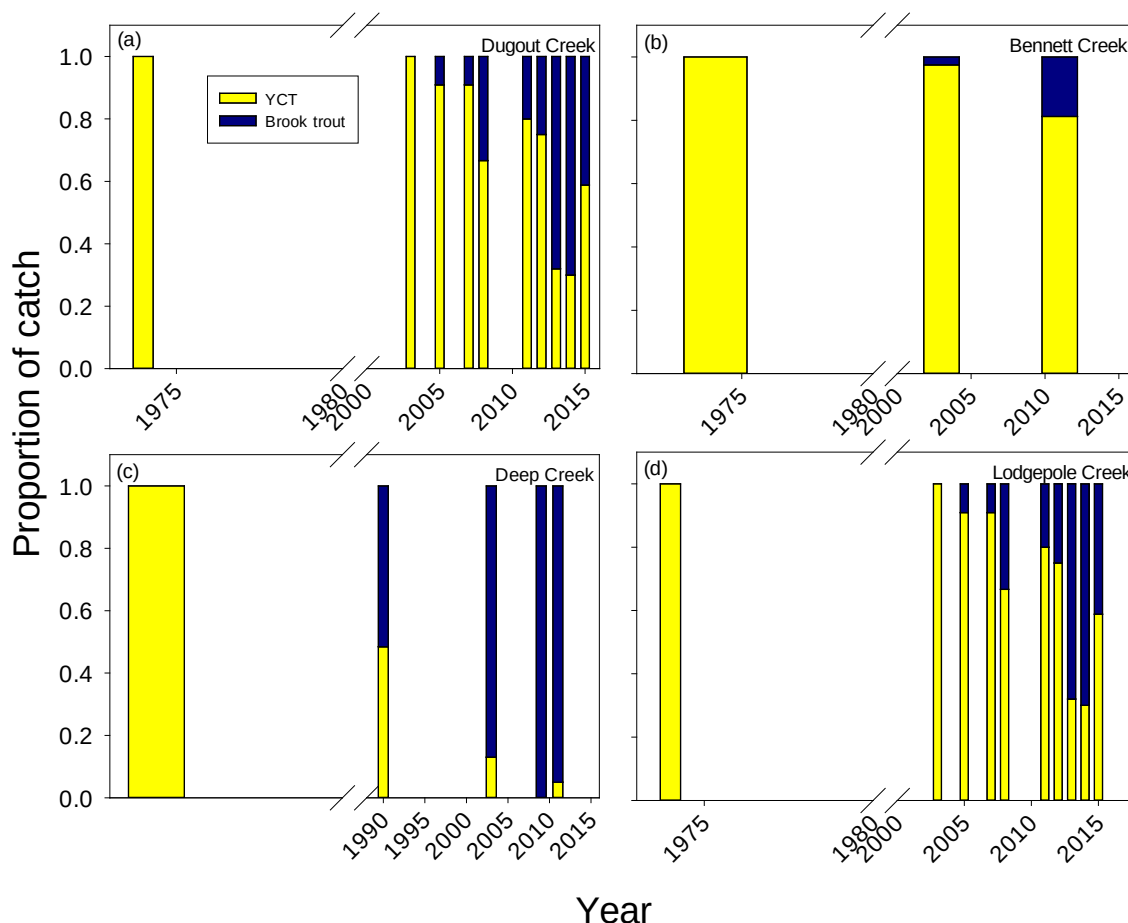


Figure 2. The proportion of catch as Yellowstone Cutthroat Trout (YCT - yellow) and Brook Trout (blue) through time at Dugout Creek (a), Bennett Creek (b), Deep Creek (c), and Lodgepole Creek (d) in the Shields River basin, Montana.

locations, particularly in the Shields River (Figure 3); the lack of juvenile cutthroat trout since the early 1990s suggest a considerable shift in the life-history expressions within the basin. The results from the Shields River differ dramatically from those observed in the Teton River, indicating the effects of nonnatives and status of extant populations of Yellowstone Cutthroat Trout are not consistent across basins within the spatial-temporal context considered herein.

Together, our results highlight the challenges of using only species distribution data and highlight the importance of long term monitoring programs. In addition, our results demonstrate the importance of using local data vs. large-scale assessments in understanding factors limiting Yellowstone Cutthroat Trout populations. We consider our results in the

context of climate change and distance to source populations of nonnative species as a means to inform conservation of native populations of Yellowstone Cutthroat Trout.

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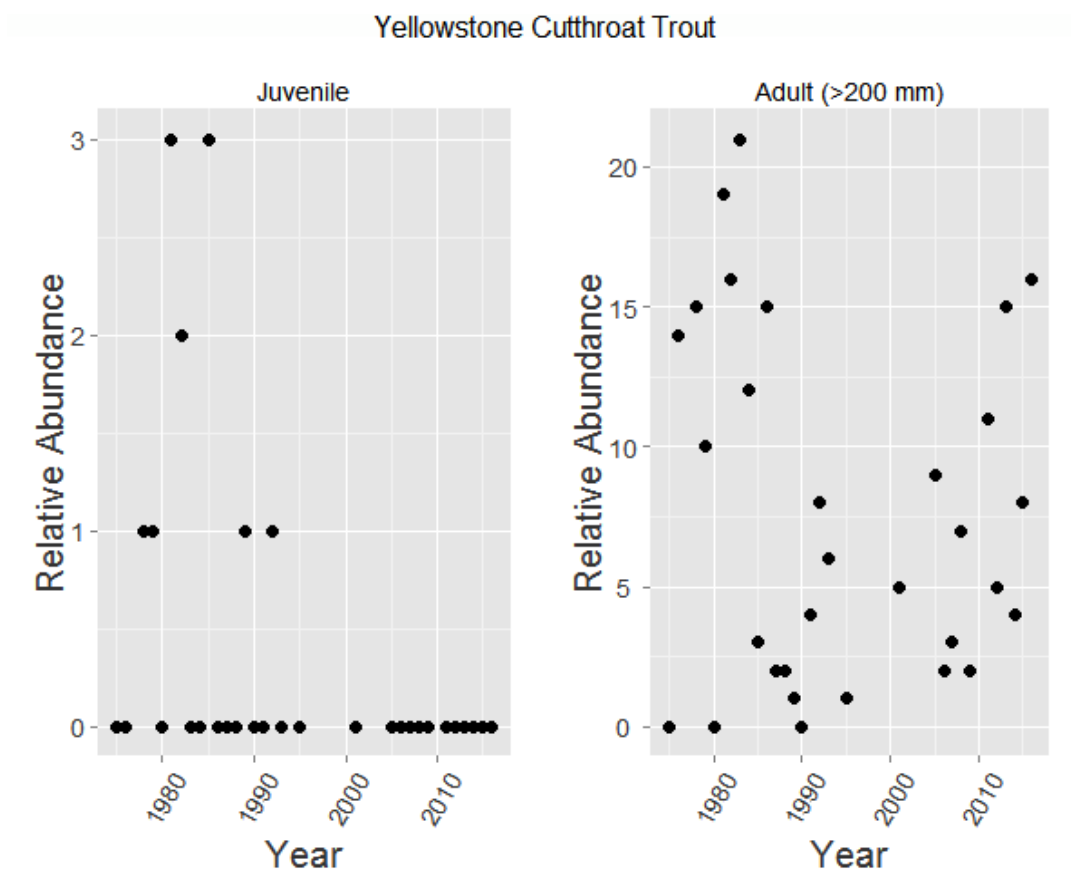


Figure 3. The relative abundance of juvenile (left panel) and adult (right panel; >200 mm) Yellowstone Cutthroat Trout at a long term monitoring site in the Shields River, Montana.



Comparing Behaviour and Habitat Preferences Between Arctic Charr and Lake Trout in a Mountain Lake

Gustav Hellström¹ & Johan Leander²

¹Department of Wildlife, Fish and Environmental Sciences, Swedish University of Agricultural Sciences, Umeå, Sweden

²Department of Ecology and Environmental Science, Umeå University, Sweden

Extended Abstract

The Lake Trout *Salvelinus namaycush* was introduced to Sweden during the 1960s, and today several viable populations exist in large deep lakes in the northern part of the country. Many of the Lake Trout populations reside in waters that also contain the native Arctic Charr *Salvelinus alpinus*, and there are concerns that Lake Trout may have a negative impact on these native fish. In this study, we tracked the behaviour and habitat use of Lake Trout and Arctic Charr in a large mountain lake using acoustic telemetry, collecting high resolution 3D positioning data on the fish over several months. The study was conducted in Lake Gautsträsk situated in the county of Västerbotten, northern Sweden (65° 55.85' N, 16° 18.12' E). This oligotrophic lake covers an area of 5.5 km² and has a mean and maximum depth of 20 and 58 m, respectively. In addition to reproducing populations of Lake Trout and Arctic Charr, the lake has native populations of large adfluvial Brown Trout *Salmo trutta*, Grayling *Thymallus thymallus*, Northern Pike *Esox lucius*, Whitefish *Coregonus lavaretus*, Bourbot *Lota lota*, and the minnow *Phoxinus phoxinus*.

Eighteen adult Lake Trout with a mean length of 720 mm (TL) were caught by angling and gill netting in the lake and in the lower part of a small tributary during the summers of 2013 and 2014. Six wild Arctic Charr (mean length 520 mm) were caught by angling in the lake in September 2014. Six additional Arctic Charr (mean length 550 mm) were taken from a nearby hatchery and stocked in the lake. All fish were tagged with acoustic transmitters (Vemco Ltd.) surgically implanted in the body cavity of the fish. The transmitters recorded both depth and temperature and transmitted a signal in an average interval of 160 s. The temperature sensors had a range from -4 to 20 °C with an accuracy of ±0.5 °C and a resolution of 0.1 °C. The depth sensor had a maximum depth of 68 m with an accuracy of ±3.4 m and a resolution of 0.3 m. A grid of 42 VR2W 69 kHz acoustic receivers (Vemco

Ltd., Halifax, Nova Scotia) was used to track the movement, depth, and temperature of the tagged fish in the lake. The detailed bathymetric map of the lake was compiled using extensive eco-sounding (Figure 1). During summer, the lake was thermally stratified with temperatures ranging from 5°C to 17°C (Figure 2). The epilimnion had temperatures between 10°C and 17°C and the thermocline was located between 17 and 20 m. Autumn circulation occurred in the last days of September, after which the lake had a homogenous temperature around 4 °C.

Lake Trout and Arctic Charr were compared with regards to temperature preference and depth use, as well as activity patterns and selection of potential spawning sites. During summer, the two species overlapped considerably in their spatial use of the lake, although Lake Trout in general resided deeper than the Arctic Charr (average 18 m vs. 12 m) (Figure 3). Lake Trout also preferred cooler temperatures than the Arctic Charr (8 °C vs 10 °C) during the summer months. Spawning was separated in time and space. Lake Trout initiated spawning earlier (end of August) and stayed at the spawning grounds for a longer period of time (2 months) compared to the Arctic Charr for which the telemetry data suggest spawning during a few days in late September. Main spawning ground for Lake Trout was at the lake outlet in shallow water (~6 m) over cliffs and large boulders, whereas Arctic Charr spawned at deeper protruding areas in the main basin of the lake (~18 m). All of the tagged Lake Trout spent considerable amount of time at the spawning grounds, compared to only 40% of the Arctic Charr. Based on the difference in behaviour and habitat utilization derived from the telemetry data, we could not determine any clear overlap between Lake Trout and Arctic Charr, suggesting that negative impacts of the introduced Lake Trout on the native Arctic Charr in our lake system may be limited.

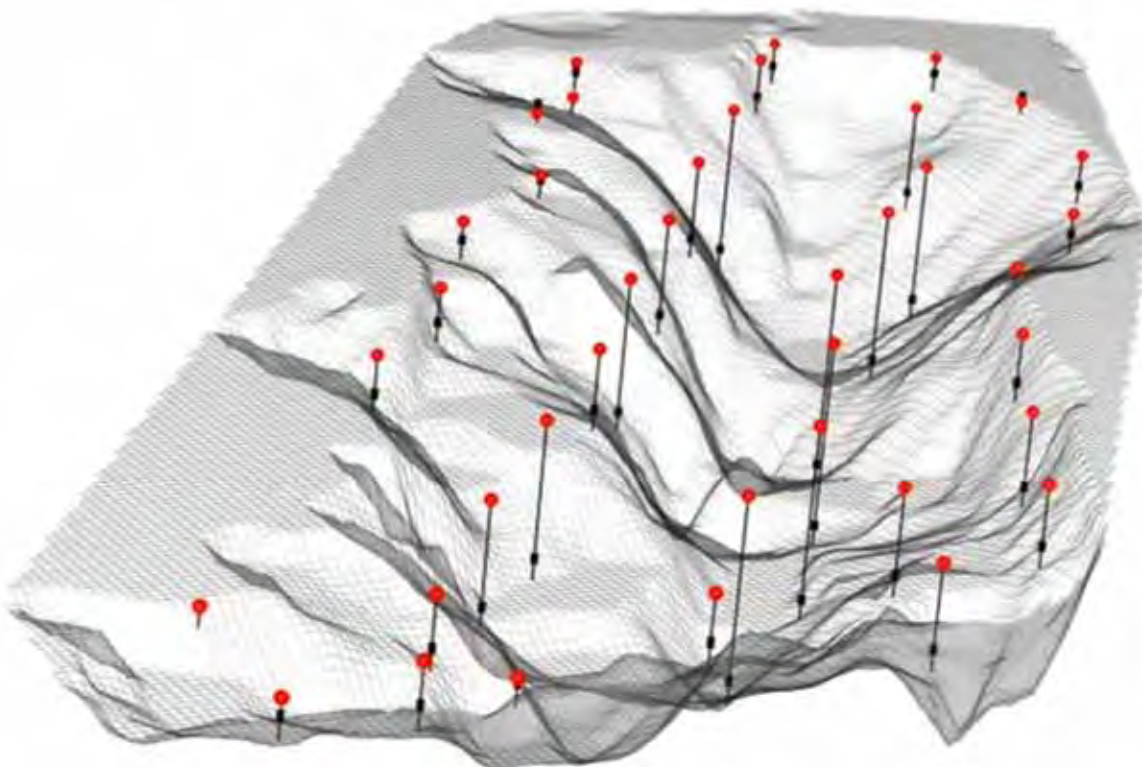


Figure 1: A bathymetric map of lake Gautsträsk, with the network of acoustic receivers illustrated as black lines with red buoys.

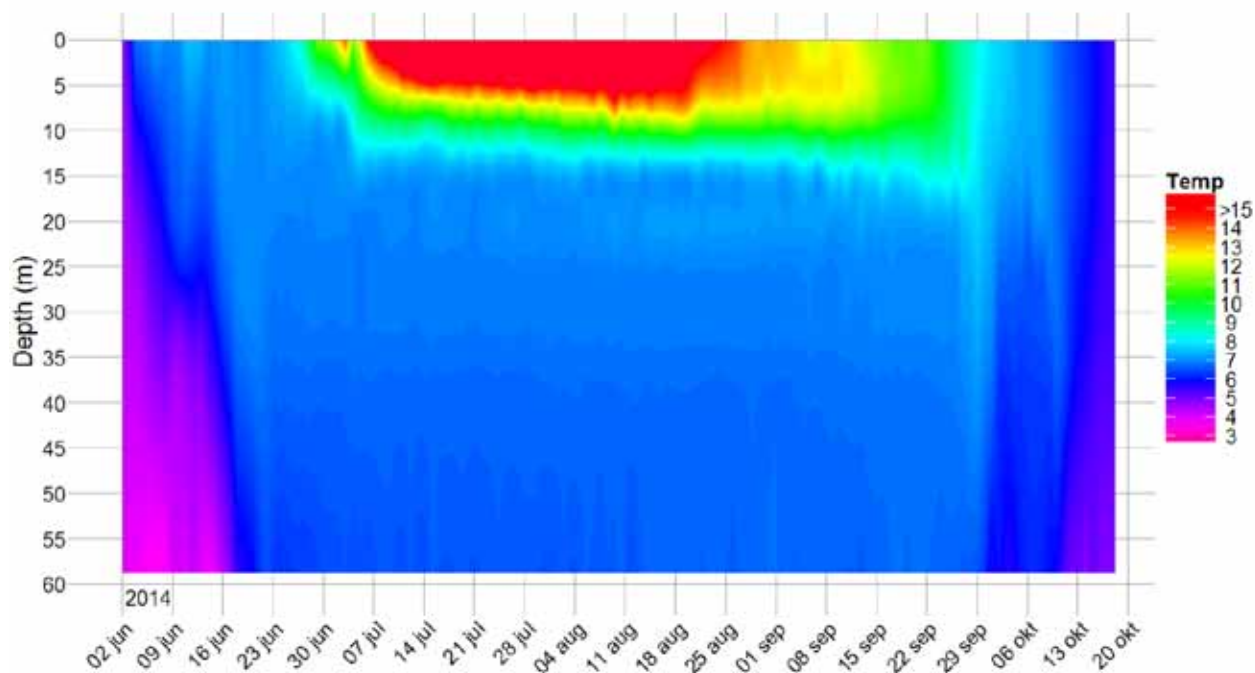


Figure 2. Thermal profile of Lake Gautsträsk interpolated from measurements recorded by temperature data loggers.

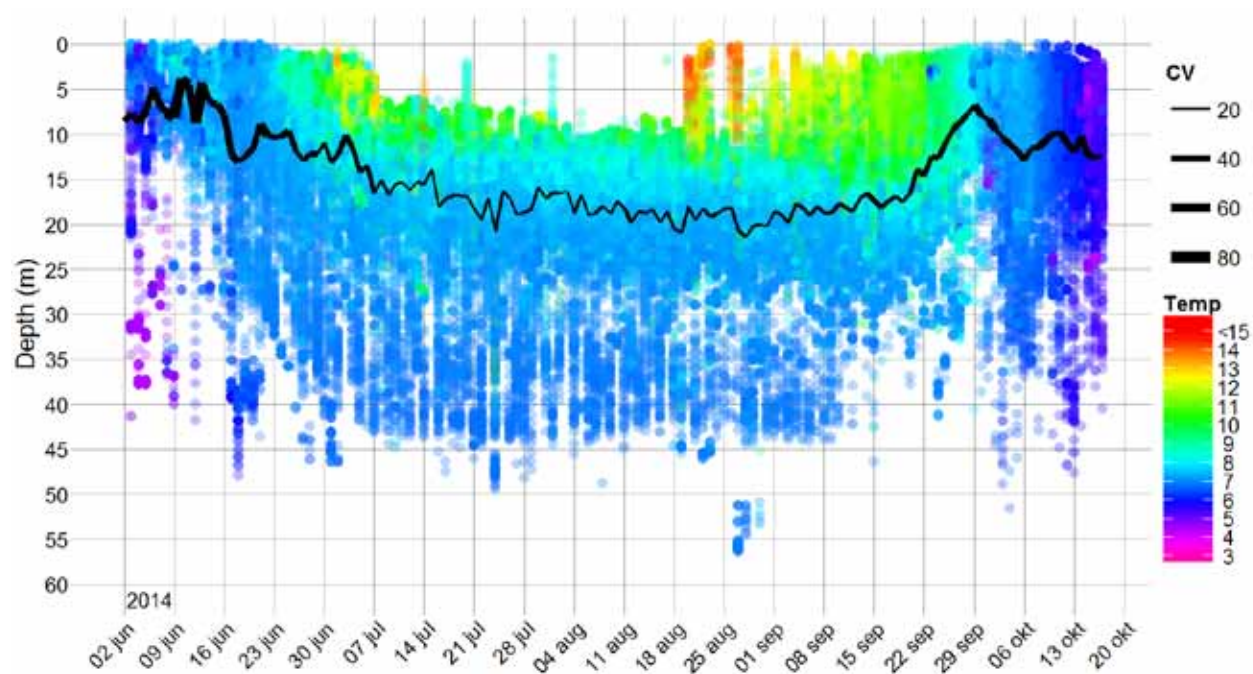


Figure 3: Bathythermal records of Lake Trout in Lake Gautsträsk. Daily mean depth is plotted with black line with thickness representing coefficient of variance (CV).



Purifying a Yellowstone Cutthroat Trout Stream by Removing Rainbow Trout and Hybrids via Electrofishing

Kevin A. Meyer¹, Patrick Kennedy², Brett High³, Matthew R. Campbell⁴

^{1,2}Idaho Department of Fish and Game, 1414 E. Locust Lane, Nampa, ID 83686 (208-465-8404)
(kevin.meyer@idfg.idaho.gov) (pat.kennedy@idfg.idaho.gov)

³Idaho Department of Fish and Game, 4279 Commerce Circle, Idaho Falls, ID 83401 (208-525-7290)

⁴Idaho Department of Fish and Game, 1800 Trout Rd., Eagle, ID 83616 (208-939-6713) (matt.campbell@idfg.idaho.gov)

Abstract—Long-term persistence of Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* in the South Fork Snake River drainage in Idaho is threatened by hybridization with introduced Rainbow Trout *O. mykiss*. We completed eight backpack electrofishing removals from 2010 to 2015 to remove Rainbow Trout and hybrids from a 9.3-km isolated reach of Palisades Creek (a South Fork tributary) to improve the purity of the population. For two removals, a subsample of *Oncorhynchus* were genetically screened at seven diagnostic nuclear DNA loci. A total of 14,092 fish were captured across all removals, of which 3,446 were putative Rainbow Trout or hybrids which were culled from the population. The proportion of the total catch comprised of Yellowstone Cutthroat Trout increased from 67% in 2010 to 86% by the second removal of 2015, whereas the proportion of Cutthroat Trout alleles increased from 80% to 90%. Considering the capture efficiencies achieved, initial hybridization levels observed, and number of removals conducted, ending phenotypic purity should have been 94% rather than 86%; this discrepancy was likely due to low capture efficiency for fish <150 mm TL, extremely high flows throughout 2011 that prevented electrofishing removals that year, and perhaps a competitive or survival advantage for Rainbow Trout and hybrids over Cutthroat Trout.

Introduction

The Rainbow Trout *Oncorhynchus mykiss* is globally one of most widely introduced species of fish outside their native range (Fausch et al. 2001), and have established self-sustaining populations on every continent but Antarctica. When they have been stocked where native salmonids exist, they often competitively displace or hybridize with the native stock of fish. For example, in the South Fork Snake River drainage in Idaho, the long-term persistence of Yellowstone Cutthroat Trout *O. clarkii bouvieri* is threatened by an increasing abundance of and hybridization with Rainbow Trout (High 2010).

Once hybridization between a native population of Cutthroat Trout and nonnative Rainbow Trout occurs, it has been argued that the process will usually lead to genomic extinction of the Cutthroat Trout parental form (Allendorf and Leary 1988). An alternate viewpoint argues that longitudinal gradients in introgression are more common in hybridized populations of Cutthroat Trout, with hybrid zones being mediated by environmental conditions and ecological differences between taxa that interact

to help segregate the parental forms and provide resistance to genomic extinction (McKelvey et al. 2016). Under either scenario, manually removing *O. mykiss* alleles will theoretically reduce both the rate and spread of hybridization and introgression in Cutthroat Trout populations (e.g., Al-Chokhachy et al. 2014).

Attempts to eradicate nonnative salmonids in streams have produced mixed results. The use of fish toxicants has sometimes been successful (e.g., Gresswell 1991), but often results in incomplete removal of target populations (reviewed in Meronek et al. 1996). Moreover, piscicides kill nontarget species as well, and biologists must consider the negative perception that application of piscicides sometimes evokes from the public (Finlayson et al. 2010). Like piscicides, electrofishing removals are also successful at reducing the abundance of nonnative species but typically do not lead to complete eradication (e.g., Thompson and Rahel 1996) unless the treatment reaches are very short (<3 km), the streams are very small (<3 m wide), and multiple electrofishing removals per year for many consecutive years are conducted (e.g., Shepard et al. 2014).

However, complete eradication of Rainbow Trout is not necessary when the goal of a project is not to completely purify a stream but rather to transform a moderate to heavily hybridized Cutthroat Trout population into a minimally hybridized population. While the value of hybridized populations of Cutthroat Trout has been intensively debated (Allendorf et al. 2004; Campton and Kaeding 2005), representatives from most fish and wildlife agencies in the western U.S. have collectively developed a Cutthroat Trout conservation strategy that includes three categories for classifying populations (UDWR 2000), including core (<1% introgressed), conservation (1-10% introgressed), and sport fish (>10% introgressed) populations. The goal of our project was to transform a heavily hybridized Cutthroat Trout population (>25% introgressed) into a minimally hybridized population (i.e., <10% introgressed).

Methods

Rainbow Trout and Hybrid Removal

Palisades Creek is a primary spawning tributary for Cutthroat Trout in the South Fork Snake River drainage in eastern Idaho. The main stem of Palisades Creek is about 30 km in length. However, about 10 km upstream from its confluence with the South Fork, there is a high-gradient, cascading section that serves as a natural barrier to fish movement, and Rainbow Trout and hybrids are absent above this barrier. In 2009, a permanent electric weir was installed on Palisades Creek about 0.7 km upstream from its confluence (High 2010), which is 90-100% efficient at stopping upstream migrating Rainbow Trout (B. High, unpublished data). The downstream electric weir and upstream natural barrier formed the boundaries of our study.

We used backpack electrofishers to capture and remove Rainbow Trout and hybrids in late summer or autumn when flows were at their annual low point. One removal was conducted in 2010 and in 2012, whereas two removals (separated by about one month) were conducted annually from 2013 to 2015. Removal efforts were not possible during 2011 because of unusually high flow conditions that rendered electrofishing inefficient and dangerous all summer and autumn. Also, high flows damaged the electric weir in 2011 early in the spawning run and it had to be shut down for the rest of the spawning period. The

weir was repaired in the winter of 2011-2012 and has been fully operational since.

Electrofishing teams consisted of 3 to 4 people operating backpack electrofishers and two or more additional people with nets and buckets. All captured trout were identified to species (hybrids were classified as a separate taxa) and measured for total length. Trout <100 mm comprised <5% of the total catch, were too small to effectively capture, and are difficult to phenotypically differentiate for these taxa, thus they were not included in our analyses, though when captured they were culled.

Taxa were separated based on previous studies (e.g., Meyer et al. 2006) which identified that, in contrast to Rainbow Trout and hybrids, Yellowstone Cutthroat Trout have (1) no white on the leading tips of the anal, dorsal, or pelvic fins, (2) fewer spots on the top of the head, (3) a bright red-orange throat slash, and (4) spots on the side of the body clustered dorsally and posteriorly. With regard to head spots, we counted spots on the head that were above the eyes, starting from just anterior of the nares and extending posterior to the point of scale formation; there is often one spot next to each nares on pure Yellowstone Cutthroat Trout, and if present these were not counted.

For six of the eight removals, we marked fish before the removal in order to estimate trout abundance using the modified Peterson mark-recapture estimator. Capture efficiency in each size group in each year was calculated as the number of marked fish caught in the recapture run divided by the total number marked.

Genetic Analyses

In 2012 and the first removal of 2015, we collected genetic samples to (1) estimate allelic purity of the population early in the project and at the end of the study, and (2) confirm the accuracy of phenotypic-based fish identification. Small fin clips were randomly collected from fish throughout the stream. We screened all samples for Rainbow Trout hybridization/introgression with seven diagnostic nuclear DNA (nDNA) markers (Occ34, Occ35, Occ36, Occ37, Occ38, Occ42 and OM55; Ostberg and Rodriguez 2002). We classified fish as Yellowstone Cutthroat Trout if they were homozygous for *O. c. bouvieri* alleles at all loci, Rainbow Trout if they were homozygous for *O. mykiss* alleles at all loci, and hybrids if they possessed alleles from both parental species. For each individual fish, using seven

codominant nDNA loci gave us a 90% probability of detecting introgression at 15% or greater.

Phenotypic introgression was calculated as the number of non-Cutthroat Trout caught in each removal divided by all *Oncorhynchus* captured. Observed levels of phenotypic introgression were compared to expectations that were approximated based on estimates of species composition and removal efficiency. For these approximations we assumed that (1) fish did not grow between removals in the same year (usually removals were separated by no more than one month), and (2) fish grew approximately 50 mm between years, up to 350 mm in length. The latter assumption is equivalent to growth rates typical for these taxa in inland Rocky Mountain streams. As an example, for the 150-200 mm size group in 2012, Rainbow Trout and hybrids comprised 28% of the catch, and capture efficiency (i.e., removal efficiency) was an estimated 38%. Based on these results and our assumed growth rates, we expected that for the next removal (i.e., the first removal in 2013), Rainbow Trout and hybrid composition would have declined to 18% of the catch in the 200-250 mm size group, instead of the 21% we observed. Expected phenotypic introgression levels could not be approximated for the smallest size group (100-150 mm) because capture efficiency for fish <100 mm was not available (i.e., we did not mark fish <100 mm).

To calculate genotypic introgression, we combined phenotypic catch data with genetic findings, because not all captured fish were genetically analyzed in the

years that genetic samples were processed, and in only two of the eight removals were genetic data analyzed. All captured fish phenotypically designated as pure Yellowstone Cutthroat Trout and pure Rainbow Trout were assigned to have 0 and 14 *O. mykiss* alleles, respectively. For fish phenotypically designated as hybrids, we calculated the mean number of *O. mykiss* alleles in the hybrids that were genetically sampled, and assumed that all the remaining hybrids not genetically sampled had the same number of *O. mykiss* alleles.

Results

We captured a total of 14,654 trout in Palisades Creek across all removals, of which 3,446 (24%) were putatively Rainbow Trout or hybrids that were removed from the stream (Table 1). An additional 200 Rainbow Trout and hybrids were caught and removed during the fish marking runs conducted during the week prior to the removals.

The phenotype matched the genotype of individual fish with a high degree of accuracy (Figure 1), especially if Rainbow Trout and hybrids were combined into one taxa (phenotypic accuracy = 94%). All phenotyping mistakes were between hybrids and parental taxa. Hybrids were mistaken for parental taxa most often when almost none or almost all of their alleles were Rainbow Trout alleles (Figure 1). Phenotypic characteristics that best distinguished Yellowstone Cutthroat Trout from hybrids were fish having (1) no white on the leading tip of the anal,

Table 1. Mark-recapture population abundance estimates and the number of Rainbow Trout and hybrids removed during each removal effort at Palisades Creek, Idaho. Population estimates were not conducted for the first and last removals. Two removals were completed in 2013-2015.

Year	Rainbow Trout and hybrid abundance						Yellowstone Cutthroat Trout abundance		
	<20 cm		20-29 cm		≥30 cm		<20 cm	20-29 cm	≥30 cm
	Estimate	Removed	Estimate	Removed	Estimate	Removed			
2010	-	260	-	426	-	148	-	-	-
2012	467	136	461	231	57	48	1,993	925	235
2013-1	1,766	294	476	56	26	13	1,953	1,820	333
2013-2	1,435	446	460	140	16	5	2,457	1,514	566
2014-1	752	148	535	169	53	24	2,322	1,054	325
2014-2	1,026	194	401	201	23	17	4,956	1,195	229
2015-1	426	97	276	163	43	23	3,554	884	215
2015-2	-	109	-	89	-	9	-	-	-

dorsal, or pelvic fins, (2) fewer than five spots on the top of the head, (3) a bright red-orange throat slash, and (4) spots on the side of the body clustered dorsally and posteriorly (Table 2).

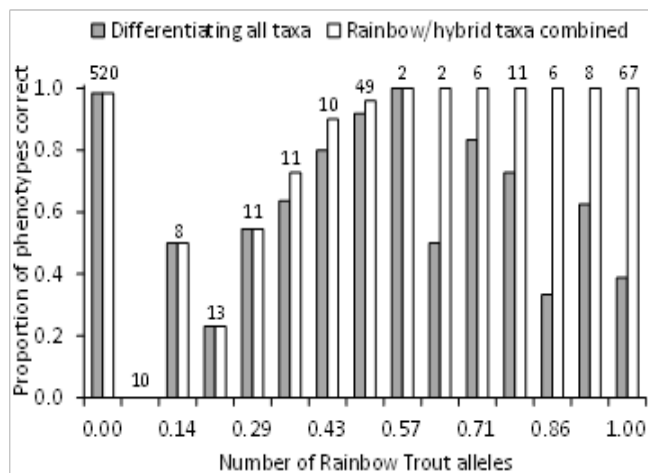


Figure 1. Accuracy of using phenotype to delineate the genotype of Yellowstone Cutthroat Trout, Rainbow Trout, and Cutthroat $\times$ Rainbow hybrids with varying proportions of Rainbow Trout alleles in individual fish in Palisades Creek, Idaho. Number of fish examined given above each set of bars.

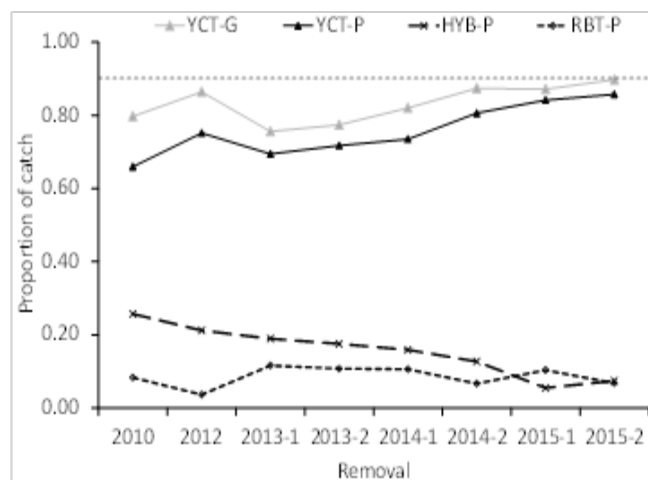


Figure 2. Proportion of the catch during eight backpack electrofishing removals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and hybrids (HYB) in Palisades Creek, Idaho. Shown for Cutthroat Trout are phenotypic (P) and genotypic (G) proportions of the catch, whereas only phenotype is depicted for Rainbow Trout and hybrids. Horizontal line (small dashes) represents the goal of at least 90% of the trout being comprised of Cutthroat Trout.

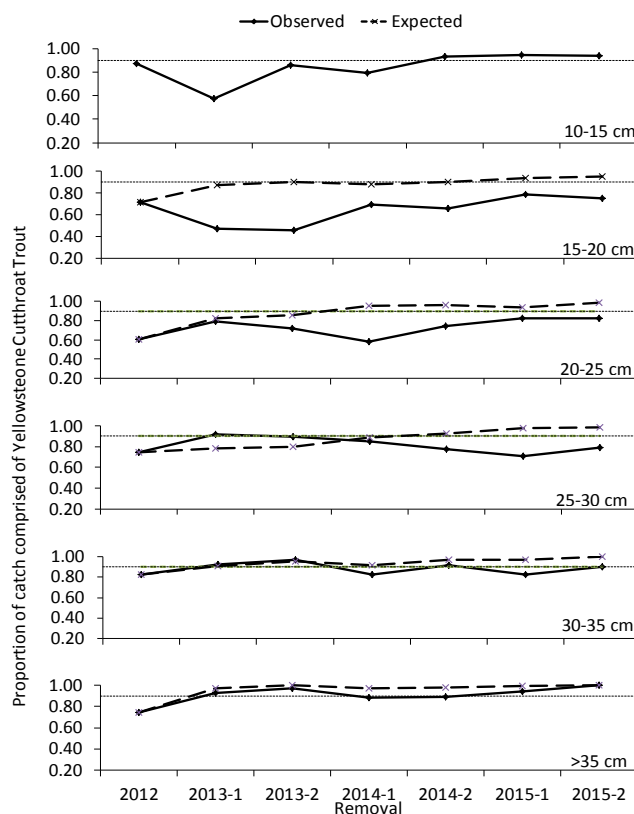


Figure 3. Observed (solid line) and expected (dashed line) proportions of captured fish by 50 mm size groups that were Yellowstone Cutthroat Trout during eight backpack electrofishing removals from 2010 to 2015 in Palisades Creek, Idaho. Horizontal lines (small dashes) represent the goal of at least 90% Cutthroat Trout.

Total trout abundance averaged 5,872 fish (range 4,138–7,830), or 631 trout/km (Table 1). Abundance increased initially for both Yellowstone Cutthroat Trout and Rainbow Trout and hybrids, but by the end of 2015, abundance since 2010 had increased by 48% for Cutthroat Trout and declined by 24% for Rainbow Trout and hybrids.

Capture efficiency for each removal averaged 38% and ranged from a low of 23% for the first removal of 2014 (the removal with the highest stream flow) to a high of 52% for the 2012 removal (when flow was the lowest of any removals). Capture efficiency generally increased as fish size increased.

The proportion of the total catch that Yellowstone Cutthroat Trout comprised increased from 67% in 2010 to 86% by the second removal in 2015 (Figure 2). Within each size class, the proportion of the total

Table 2. Frequency of occurrence of the variations in phenotypic characteristics used to distinguish Yellowstone Cutthroat Trout from Rainbow Trout and hybrids in Palisades Creek, Idaho.

		Genotypic composition (%)			
		Yellowstone			
Meristic	Characteristic	Cutthroat	Hybrids		Rainbow
		Trout	>F ₁	F ₁	Trout
Head spots	Five or more	4.6	68.2	91.2	96.3
	Fewer than five	95.4	31.8	8.8	3.7
Belly	Orange hue	26.1	9.1	5.9	0.0
	White hue	73.9	90.9	94.1	100.0
Pelvic fins	White tips present	0.5	52.3	58.8	100.0
	White tips absent	99.5	47.7	41.2	0.0
Anal fin	White tips present	0.9	56.8	79.4	100.0
	White tips absent	99.1	43.2	20.6	0.0
Dorsal fin	White tips present	0.0	38.6	70.6	88.9
	White tips absent	100.0	61.4	29.4	11.1
Throat slash	Bright red-orange and prominent	89.9	34.1	17.6	0.0
	Dull but continuous	4.1	11.4	17.6	0.0
	Faint and barely visible	6.0	43.2	64.7	59.3
	Absent	0.0	11.4	0.0	40.7
Body spots	Smaller; distributed evenly on sides	3.7	70.5	76.5	100.0
	Larger; clustered dorsally and posteriorly	96.3	29.5	23.5	0.0
Side coloration	Presence of reddish hue	5.5	61.4	85.3	96.3
	Absence of reddish hue	94.5	38.6	14.7	3.7

catch comprised of Yellowstone Cutthroat Trout varied through time (Figure 3). For the smallest size classes (100-200 mm), Cutthroat Trout comprised 75% of the total catch in 2010 and 88% by the second removal of 2015, constituting a 17% increase. In comparison, for intermediate-sized fish (200-299 mm) Cutthroat Trout comprised 60% of the catch in 2010 and 81% by the second removal of 2015 (a 35% increase), and for spawning-sized fish (≥ 300 mm), Cutthroat Trout comprised 60% of the catch in 2010 and 91% by the second removal of 2015 (a 52% increase).

For small and medium size classes of fish, the decline in the proportion of the total catch comprised of Rainbow Trout and hybrids did not keep pace with expectations, whereas for the largest size classes, expectations more closely matched the observed reduction in Rainbow Trout and hybrid composition in the population (Figure 3). A large group of Rainbow Trout and hybrids 100-200 mm in length (mostly age-1 and age-2 fish) was apparent in both 2013 removals

(Table 1; Figure 3), which in subsequent years was also apparent in larger size classes, suggesting that 2011 was a successful spawning year for Rainbow Trout and hybrids. By the end of the study, based on initial fish composition and our estimated capture efficiency for each size class during each removal, phenotypic purity for the entire Cutthroat Trout population should have reached 94%, rather than the 86% we observed.

In 2010, across all size classes, genetic analyses indicated that 80% of the alleles were Cutthroat Trout alleles, compared to 67% of the fish being phenotypically classified as Cutthroat Trout. By the final removal effort of 2015, an estimated 90% of the alleles were Cutthroat Trout compared to 86% of the fish. The gap between the number of Cutthroat Trout alleles and fish in Palisades Creek narrowed from the start to the end of the study because the overall abundance of hybrids declined to a greater extent (83% reduction) than did Rainbow Trout abundance (51%; Figure 2).

Purification of the stream worked better in the upper reaches of the stream relative to the lower reaches (Figure 4). In fact, for the smallest size class in the lowest reach, the proportion of the total catch that Yellowstone Cutthroat Trout comprised actually declined through time.

Discussion

The capture efficiencies we achieved and the number of removals we conducted should have reduced the number of non-Cutthroat Trout fish in Palisades Creek to 6% instead of the 14% level that we observed. We believe that one or more of the following issues may have diminished the observed success of the removals relative to expectations. First, this was partly an artifact of our inability to capture small fish, with capture efficiency for fish 100–150 mm in length averaging only 0.11, and for fry being inherently even lower (though we did not empirically measure it). Moreover, extremely high flows throughout 2011 prevented weir operation, and precluded electrofishing removals that year, so that age-0 and age-1 Rainbow Trout and hybrids at the start of the study were not appreciably vulnerable to removal for the first several years of the study. Unusually high flows in 2011 theoretically should have diminished spawning success of Rainbow Trout and hybrids (Fausch et al. 2001) because they spawn in later winter to early spring and fry would have emerged during the height of the spring flooding, but

catch data in later years indicated that 2011 was in fact a productive year for *O. mykiss* recruitment.

Another explanation for why expectations lagged behind observed reductions in *O. mykiss* alleles is that the weir on Palisades Creek may not completely block upstream migrating Rainbow Trout and hybrids during operation, or may not be operated long enough each year to block all Rainbow Trout and hybrid spawners attempting to access Palisades Creek. The fact that purification was least effective at the bottom of the study area supports the reinvasion supposition, but the following observations do not: (1) weir efficiency is monitored annually and is usually 90–100% during operation (B. High, unpublished data); (2) the proportion of the catch at the weir comprised of Rainbow Trout and hybrids is already very low in most years (about 6% on average; B. High, unpublished data); (3) the gap in time between the start and end dates of weir operation, and the first and last Rainbow Trout or hybrid caught in the trap each year, indicates that the weir is annually being operated well before and after the spawning run of Rainbow Trout and hybrids occurs; and (4) although the electrical component of the weir is annually deactivated in mid-July, the fish trap is closed when the weir is not being operated, and there is little to no pool for jumping over the waterfall at the weir, so even during the off-season the weir is largely if not completely a barrier to upstream movement. Another potential source of stream recolonization (besides fish getting past the weir) is the few man-made ponds on private property adjacent to the lower 3 km of the treatment reach. In Idaho, IDFG oversees private pond fish stocking regulations and enforcement, and in our study area, any Rainbow Trout stocked on private property must be sterile, though such regulations may be violated by landowners. Private ponds are also required to be screened to prevent fish escapement, and in order to create immediate fisheries, they are usually stocked with catchable-sized fish, which are readily apparent at capture (based on the condition of their fins) but were never encountered during our study. Taken collectively, we deem it unlikely that stream recolonization by any of these means was occurring at a sufficient rate to explain much of the lag in *O. mykiss* allele reduction in Palisades Creek.

One final possibility is that Rainbow Trout and hybrids may have an interspecific competitive or selective advantage over Yellowstone Cutthroat

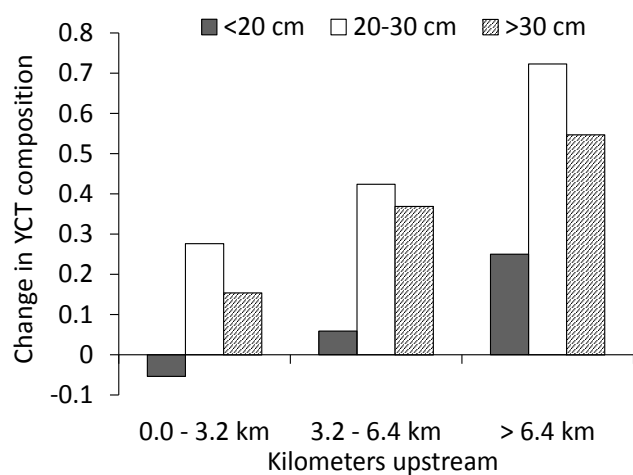


Figure 4. Longitudinal differences in the change (from the first to the last removal effort) in the proportion of the trout population comprised of Yellowstone Cutthroat Trout in Palisades Creek, Idaho.

Trout. For Cutthroat Trout, a competitive disadvantage with nonnative Brook Trout *Salvelinus fontinalis* is fundamentally acknowledged (see review in Dunham et al. 2002), but with Rainbow Trout, competition is considered less important in the extirpation of Cutthroat Trout populations than introgression (Young 1995). Nevertheless, Rainbow Trout and hybrids tend to spawn earlier than Yellowstone Cutthroat Trout (Henderson et al. 2000), giving juvenile fish the additional advantage of larger size stemming from earlier emergence. Results from stochastic Lotka–Volterra modeling applied to long-term population monitoring in the South Fork Snake River suggest that hybridization has been the primary mechanism for reductions in Yellowstone Cutthroat Trout abundance in the river, but direct competition was supported by the models as well (Van Kirk et al. 2010). While some studies suggest that hybrids have reduced fitness compared to parental Westslope Cutthroat Trout *O. c. clarkii* (Muhlfeld et al. 2009), this may not hold true for Yellowstone Cutthroat Trout. Regardless of which of these explanations contributed to the slight lag in purification we observed, we found that after eight removals over five years, the goal of reducing Rainbow Trout and hybrids to $\leq 10\%$ in Palisades Creek, at least at the level of *O. mykiss* alleles in the entire *Oncorhynchus* population, was nevertheless achieved.

We observed a stronger response to removal of Rainbow Trout and hybrids in upstream reaches of the study area compared to downstream reaches. Since the weir prevents reinvasion of Rainbow Trout and hybrids downstream of the study area, the most likely explanation for the longitudinal gradient we observed in removal response is immigration by Yellowstone Cutthroat Trout from upstream of the natural velocity barrier. Since Rainbow Trout and hybrids are absent upstream of the barrier, any influx of Cutthroat Trout at the upper end of our study area would have biased the computed rate of Rainbow Trout reduction relative to lower reaches of the stream.

Conclusions

While others have shown that removing Rainbow Trout and hybrids from portions of hybridized Cutthroat Trout populations has led to reduced levels of introgression in the population (Al-Chokhachy et al. 2014), our study is the first we are aware of attempting to reduce *O. mykiss* alleles throughout an entire Cutthroat Trout population. For the Yellowstone Cutthroat Trout population in Palisades Creek, eight

electrofishing removals were needed to reduce the percentage of fish with *O. mykiss* alleles from 33% to 14% and the percentage of *O. mykiss* alleles in the population from 20% to 10%. Completely purifying a hybridized Yellowstone Cutthroat Trout population using electrofishing suppression will be difficult, but not impossible if fitness selects against Rainbow Trout alleles (Muhlfeld et al. 2009).

A number of factors should be considered by biologists attempting such a project. Of utmost importance is that the target Cutthroat Trout population be isolated from future Rainbow Trout reinvasion by a downstream barrier. Target populations would ideally reside in streams where high electrofishing capture efficiency (i.e., at least 25–30%) can be achieved so that a meaningful percentage of Rainbow Trout and hybrids can be removed with each electrofishing pass. Biologists must be able to differentiate Cutthroat Trout, Rainbow Trout, and their hybrids with a high degree of accuracy to avoid inadvertently culling Cutthroat Trout or releasing captured hybrids; in our study only 0.5% of the Yellowstone Cutthroat Trout that were captured were mistakenly identified as hybrids, and only 41 of the 897 Rainbow Trout alleles detected in the fish we analyzed genetically were accidentally released due to mistaken identification. If the goal is not complete purification, biologists should recognize that hybridization within Cutthroat Trout populations is a dynamic process influenced by demography, zoogeography, and climate, and is rarely at equilibrium (McKelvey et al. 2016). As such, maintaining the proportion of *O. mykiss* alleles in the treated population below the targeted level may require periodic maintenance electrofishing removals, as has been recommended for nonnative Brook Trout when their complete eradication from Cutthroat Trout streams is not possible (Peterson et al. 2008). Finally, periodic monitoring of genotypic introgression levels in the population should be undertaken to confirm any assessments based on phenotypic characterization.

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Improving a High Mountain Lake Fishery by Stocking Tiger Muskellunge to Control an Overabundant Wild Brook Trout Population

Jordan Messner

Regional Fisheries Biologist, Idaho Department of Fish and Game, 99 HWY 93 N, Salmon, Idaho, 83467, phone: 208-756-2271, Jordan.messner@idfg.idaho.gov

Abstract—Naturally-reproducing trout populations in high mountain lakes (HMLs) do not always provide desirable sport fisheries. Overabundance can often lead to poor growth rates, which can be improved if abundance is reduced to increase forage availability. In an effort to reduce abundance and improve size structure of an overabundant wild Brook Trout *Salvelinus fontinalis* population in a HML in central Idaho, we stocked Tiger Muskellunge *Esox lucius* x *Esox masquinongy* (TM) on three occasions over a 15- year period. Brook Trout abundance was significantly reduced immediately following each stocking event, but increased again 3-4 years following each event. In years when Brook Trout abundance was reduced, size structure improved significantly. From 2002 to 2016, mean Brook Trout TL increased 71 mm, mean relative weight increased from 78 to 95, and proportion of Brook Trout ≥ 250 mm increased from 0.07 to 0.78. In addition to improving Brook Trout size structure, TM showed excellent growth and provided a unique opportunity for anglers to catch trophy-size fish. These results show that, under the right conditions, stocking TM at low density in HMLs with overabundant, stunted trout populations, can improve overall fishery quality.

Introduction

High mountain lake (HML) trout fisheries are highly regarded as resources that can offer unique angling experiences in remote and scenic backcountry areas. The Idaho Department of Fish and Game (IDFG) estimates that more than 40,000 anglers fish HMLs in Idaho each year, making this resource an important contributor to the state's recreational economy. Historically, around 95% of HMLs in the western U.S. were fishless prior to the 19th century (Donald 1987; Bahls 1992). In Idaho, much like other western states, HML stocking increased exponentially with the aid of aircrafts after World War II, and IDFG initiated a formal HML stocking program in 1949. Although many HMLs in Idaho still require stocking to support recreational sport fisheries, perhaps as much as 30% of the more than 3,000 HMLs in Idaho currently support self-sustaining wild trout populations.

While wild trout populations in HMLs have the benefit of providing fishing opportunity without the aid of stocking, they do not always provide desirable sport fisheries (Walters and Vincent 1973; Donald et al. 1980; Donald and Alger 1989). For example, wild Brook Trout *Salvelinus fontinalis* populations in HMLs throughout the western US often exhibit high rates of natural reproduction, early age-at-maturity,

and low rates of mortality, thus are prone to becoming overabundant and “stunted” (Donald et al. 1980; Hall 1991; Johnson et al. 1992). Stunted fish populations exhibit poor growth rates due to a lack of adequate forage, resulting in low maximum size attainment, and poor body condition. Wild trout populations like these are often not desirable to anglers (Donald et al. 1980), so fisheries managers look for solutions to improve their quality. In some cases, if fish abundance is reduced enough to increase forage availability, growth can be significantly improved (Donald and Alger 1989; Hall 1991). In other cases, complete elimination of the population may be desired in order to restock the lake with alternative species and increase fishery quality (Klein 1960; Walters and Vincent 1973).

A variety of methods can be effective at reducing the abundance of or eliminating undesirable wild trout populations in HMLs. Physical removal with gill nets has proven effective in some smaller lakes (<3 ha), but can take many years to complete (Knapp and Matthews 1998; Parker et al. 2001). Chemical treatment with piscicides (e.g., rotenone) can be very effective for full eradication (Walters and Vincent 1973), but can cause deleterious effects to native, non-target, flora and fauna (Finlayson et al. 2010). Electrofishing can be an effective method for removal in streams (Shepard et al. 2014), but is generally impractical in HMLs. A more passive approach is to

stock sterile predators as a biological control to reduce fish abundance (Hoddle 2002; Koenig et al. 2015). Beginning in 1998, IDFG began experimenting with the use of Tiger Muskellunge *Esox lucius* $\times$ *Esox masquinongy* (TM) to reduce the abundance of or eliminate undesirable wild Brook Trout populations in HMLs. Koenig et al. (2015) later investigated this approach further by stocking TM in 13 HMLs in Idaho, and found that Brook Trout abundance was greatly reduced in nearly all lakes, and Brook Trout were completely eliminated from four study lakes within 5 years of stocking. In this study, our objective was to determine if repeated introductions of TM at a low stocking density, and at periodic intervals, could significantly improve Brook Trout size structure and increase fishery quality over a long-term period at Carlson Lake, in central Idaho.

Methods

Carlson Lake (WGS84 datum: 44.28153°N, 113.75283°W) is a subalpine lake approximately 3.5 ha in size located in the Pahsimeroi River drainage at 2,438 m elevation. Subterranean flow from the lake drains into Double Springs Creek, a tributary of the Pahsimeroi River (Upper Salmon River Basin), but there is essentially no outlet and the inlet flow is seasonally intermittent. Carlson Lake has a highly vegetated littoral zone that extends for an average of approximately 12 m from shore and averages around 1 m deep, around the entire perimeter of the lake. Numerous spring upwellings occur in the littoral region of the lake.

TM were reared at IDFG's Hagerman State Fish Hatchery to an average size of approximately 300-330 mm TL, and were stocked in Carlson Lake in June of 2002 (density = 11.7 fish/ha), 2006 (9.1 fish/ha), and 2013 (20.0 fish/ha), following Brook Trout sampling in those years.

The Brook Trout population in Carlson Lake was sampled prior to the introduction of TM in 2002 to collect baseline abundance and size structure information, and periodically (i.e., every 1 to 3 years) throughout the duration of this study (2002 to 2017) to assess their response to TM introductions. Brook Trout were sampled on all occasions using paired floating and sinking experimental gill nets that were 64 m long and 2 m deep, with six different mesh sizes (19, 25, 32, 38, 51, and 64 mm bar mesh). Catch-per-unit-effort (CPUE), in units of fish/hr of gill net set, was used as

an indication of the relative abundance of Brook Trout in the lake.

Prior to gill netting in 2002, 2005, 2013, and 2015, we marked Brook Trout to estimate population abundance using the Peterson mark-recapture method with the Chapman modification (Ricker 1975). We then calculated 95% confidence intervals (CIs) around abundance estimates and examined estimates for CI overlap to detect significant differences.

Brook Trout captured in gill nets during all sampled years were counted, measured (total length to the nearest mm), and weighed (nearest g). All TM captured from 2003 to 2017 were counted and measured, but were only weighed in 2003, 2016, and 2017.

We built length-frequency histograms for Brook Trout each year to examine trends in overall size structure, and calculated the proportion of Brook Trout ≥ 250 mm to describe trends in the relative proportion of "quality size" fish present in the population. Significant changes in the proportion of Brook Trout ≥ 250 mm between years were those in which 95% CIs around the difference between proportions did not overlap zero (Fleiss et al. 1981).

For all fish captured and weighed, we calculated relative weights (W_r) to compare overall change in body condition throughout the study period. Standard weight (W_s) was calculated using intercept and slope values for Brook Trout ($a = -5.186$, $b = 3.103$; Hyatt and Hubert 2001) or TM ($a = -6.126$, $b = 3.337$) (Rogers and Koupal 1997), then W_r was calculated for each fish:

$$\text{Log}_{10}(W_s) = a + b * \text{Log}_{10}(\text{total length (mm)})$$

$$W_r = \left(\frac{\text{weight (g)}}{W_s} \right) * 100$$

ANOVA with post hoc Tukey's test for pairwise comparisons was used to test for differences in mean length between years, and relative weight between study years. CIs were calculated for mean relative weights according to methods outlined in Murphy et al. (1990).

Otoliths were removed from a subsample of the Brook Trout caught (about 5 per 10 mm size class) in 2006 and 2015 for age and growth analysis. Ages were also estimated for all captured TM using a combination of otoliths, size at the time of capture, and the presence/absence of PIT tags implanted in all TM stocked in 2013. All otoliths were mounted

in epoxy and cross-sectioned using an isometric saw (Beamish 1979). Sections were digitized under 40x magnification and read by two independent readers. If independent readers were not in agreement on an age, a third reader was used to assign an age to the otolith. Using the program Fishery Analyses and Modeling Simulator (Slipke and Maceina 2001), we constructed two age-length keys for Brook Trout (2006 and 2015) in order to determine the proportion of fish of each age, in each length group. Mean length-at-age was then calculated from the age-length keys according to the methods outlined in Murphy and Willis (1996). To estimate growth rates of the Brook Trout populations for each year (2006 and 2015) we solved for the von Bertalanffy growth parameters (t_0 , k , and L_∞), and constructed growth curves for both years, from age-2 to age-7, for comparison, using the equation:

$$L_t = L_\infty (1 - e^{-k(t-t_0)})$$

where:

L_∞ = maximum theoretical length (length infinity) that can be attained;

k = growth coefficient;

t = time or age in years;

t_0 = time in years when length would theoretically be equal to zero and;

e = exponent for natural logarithms

Results and Discussion

We estimated approximately 9,000 age-2 and older Brook Trout (95% CI = 7,474 – 11,064), or 2,570 fish/ha, were in Carlson Lake prior to introducing 40 TM in 2002. One year later, Brook Trout relative abundance (CPUE) declined 63%, from 3.69 fish/hr to 1.35 fish/hr (Table 1, Figure 1). However, abundance began to increase again three years later. CPUE increased to 1.62 fish/hr in 2005, and we estimated Brook Trout abundance at approximately 6,100 fish (4,196 – 9,262), or 1,740 fish/ha. CPUE increased again in 2006, to 2.32 fish/hr (Table 1, Figure 1), and we decided to stock another 32 TM. After stocking in 2006, Brook Trout CPUE declined again, to 1.62 fish/hr in 2009. Abundance may have been reduced even further immediately following stocking, but the lake was not surveyed in 2007 or 2008. Similar to what was observed with the first TM introduction, abundance reductions were not long-lived and CPUE increased again within five years after stocking (to 2.16 fish/hr in 2011 and 4.78 fish/hr in 2013). By 2013, Brook Trout abundance was estimated at approximately 10,900 fish (9,182 – 13,008), or 3,120 fish/ha, and CPUE was the highest we observed throughout the entire study period (Table 1, Figure 1). We introduced TM again in 2013, at twice the stocking density of previous events ($n = 70$), and saw a 70% decline in Brook Trout CPUE (to 1.44 fish/hr) and

Table 1. Brook Trout relative abundance (CPUE) and size structure (mean TL mm, mean relative weight W_r , and proportion ≥ 250 mm TL) throughout the study period (2002 - 2017) at Carlson Lake.

Year	Relative abundance			Size structure		
	Gill net effort (hrs)	# caught	CPUE (fish/hr)	Mean TL (range) (mm)	Grand mean W_r ($\pm$ 95% CI)	Prop ≥ 250 mm TL ($\pm$ 95% CI)
2002	147.8	546	3.69	201 (109-276)	78 ($\pm$ 0.8)	0.07 ($\pm$ 0.02)
2003	416.9	562	1.35	209 (96-270)	59 ($\pm$ 2.3)	0.06 ($\pm$ 0.03)
2005	369.5	599	1.62	231 (145-290)	89 ($\pm$ 1.8)	0.48 ($\pm$ 0.08)
2006	64.8	150	2.32	216 (127-301)	104 ($\pm$ 2.5)	0.47 ($\pm$ 0.08)
2009	151.7	246	1.62	234 (136-312)	87 ($\pm$ 2.0)	0.45 ($\pm$ 0.07)
2011	132.7	287	2.16	218 (115-291)	80 ($\pm$ 1.3)	0.26 ($\pm$ 0.05)
2013	172.5	825	4.78	220 (150-292)	75 ($\pm$ 0.5)	0.32 ($\pm$ 0.03)
2015	75.0	108	1.44	252 (165-289)	86 ($\pm$ 1.5)	0.81 ($\pm$ 0.08)
2016	82.7	67	0.81	272 (169-351)	95 ($\pm$ 2.1)	0.78 ($\pm$ 0.09)
2017	84.2	184	2.23	234 (151-397)	94 ($\pm$ 1.9)	0.39 ($\pm$ 0.08)

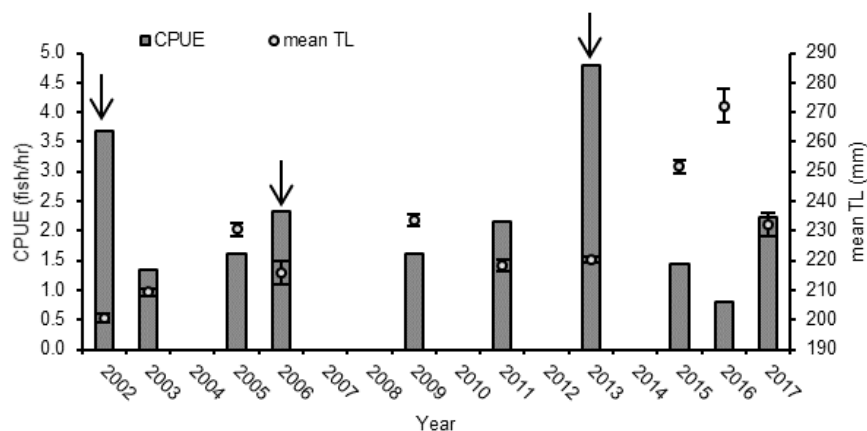


Figure 1. Relative abundance (CPUE) and mean TL (mm) ($\pm$ 95% CIs) of Brook Trout sampled at Carlson Lake during the study period (2002 - 2017). $\downarrow$ denotes years when Tiger Muskellunge were stocked, after Brook Trout were sampled.

another significant reduction in overall abundance to an estimated 2,680 fish (1,653 – 4,748), or 765 fish/ha. Brook Trout CPUE continued to decline to a study period low of 0.81 fish/hr in 2016, but again increased in 2017 (2.23 fish/hr), four years after the last TM stocking event.

Across the 15-year study period, mean Brook Trout length and proportion of Brook Trout ≥ 250 mm were negatively correlated to CPUE ($R^2 = 0.49$ and 0.31 , respectively; Figure 2). Age and growth data collected in 2006 and 2015 suggests that in years of

relatively low CPUE (e.g., 2015), proportion of age-2 and age-3 Brook Trout in the lake was much lower than in years when abundance was relatively higher (e.g., 2006). This can be seen in all years immediately following TM stocking events, where younger size/age classes of Brook Trout were nearly absent from our catch (Figure 3). Relative length-frequencies of Brook Trout sampled throughout the study period indicate that TM preyed heavily upon Brook Trout < 180 mm, especially in the first few years immediately following stocking (Figure 3). This likely increased

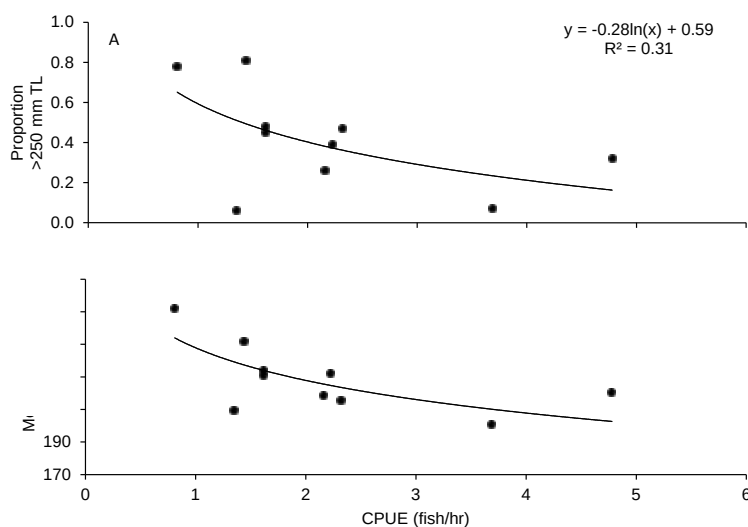


Figure 2. Logarithmic relationships between CPUE (fish/hr) and (A) proportion of Brook Trout ≥ 250 mm and (B) mean TL observed during study years (2002 - 2017), at Carlson Lake.

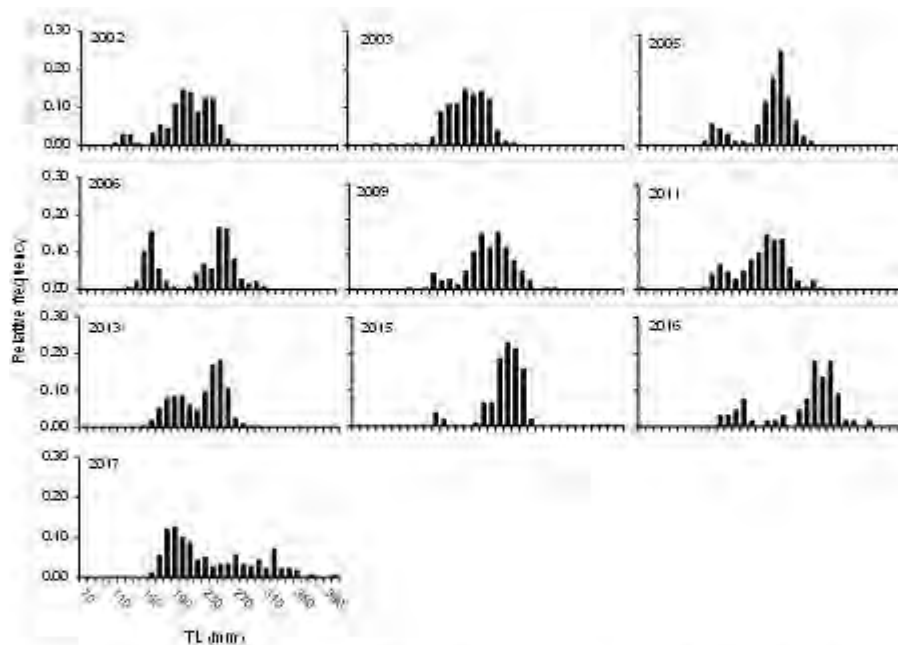


Figure 3. Relative frequencies of Brook Trout captured, by TL, in Carlson Lake during the study period (2002 - 2017). ¹Denotes years when Tiger Muskellunge were stocked, after Brook Trout were sampled.

growth rates for younger Brook Trout that avoided predation (Figure 4), thereby leading to significant improvements in overall size structure (mean TL, mean W_r , and proportion ≥ 250 mm) during the study period.

ANOVA results showed significant differences in both mean TL ($F = 58.5$, $df = 9$, $p < 0.01$) and mean W_r ($F = 200.3$, $df = 8$, $p < 0.01$) among study years. Both metrics generally improved with reductions in abundance, and declined with subsequent increases in abundance as mentioned earlier. Mean Brook Trout length increased significantly ($p < 0.01$) from 2002 to 2005, following the first introduction of TM, but declined again in 2006 ($p = 0.06$) with an increase in CPUE (Table 1, Figure 1), and an increase in the relative frequency of younger Brook Trout present in our sample (Figure 3). Mean Brook Trout length increased significantly ($p < 0.01$) again following the second TM introduction, to 234 mm in 2009 (Table 1, Figure 1), and the maximum length observed increased to 312 mm (Table 1). However, by 2013 all size structure metrics once again declined with increases in CPUE (Table 1). With the third stocking event in 2013, a dramatic reduction in Brook Trout abundance resulted in significant improvements to size structure once again (Table 1, Figure 1). Mean Brook Trout

length increased significantly ($p < 0.01$) to 272 mm by 2016, mean relative weight increased significantly ($p < 0.01$) to 95, and proportion of Brook Trout ≥ 250 mm increased significantly to 0.78, compared to only 0.07 in 2002 (Table 1). By 2017, mean TL and proportion ≥ 250 mm declined as a result of an increase in the relative frequency of smaller, younger Brook Trout in the sample (Table 1, Figures 1 and 3), but maximum TL increased to 397 mm; much higher than anything we had previously observed.

Based on Brook Trout length-frequencies and ageing data collected throughout the study period, TM preyed heavily upon age-3 and younger Brook Trout in the first few years immediately following each stocking event (Figures 3 and 4). Those heavy predation rates resulted in excellent growth for TM stocked in Carlson Lake throughout the study period, and especially during the first few years after each stocking event (Table 2). Within the first three years in the lake, TM stocked in 2002 grew an average of 144 mm/year, those stocked in 2006 grew an average of 157 mm/year, and those stocked in 2013 grew an average of 125 mm/year (Table 2). Growth slowed slightly after year three for all stocked groups, but body condition remained excellent for all TM captured throughout the course of the study (grand mean

$W_r = 113$; range, 96 – 135; Table 2). The largest and oldest TM we captured in the lake during the 15-year study period was 1,067 mm at age-12; weight was not measured (Table 2).

Introductions of TM in Carlson Lake undoubtedly improved overall fishery quality throughout the study period. Not only did Brook Trout size structure improve as a result of TM reducing their abundance, but excellent growth of TM resulted in a unique opportunity for anglers to catch trophy-size fish. Setting minimum length harvest restrictions for TM at Carlson Lake (none under 1,016 mm) allowed predators to act as biological controls for at least 7 years before being susceptible to harvest, while at the same time providing a trophy catch-and-release fishery. Improvements in Brook Trout size structure would not have been sustained over the long-term without protective regulations and periodic introductions of TM throughout the study period.

Koenig et al. (2015) found that the ability of TM to eradicate Brook Trout was likely a function of habitat suitability and survival after stocking, and that repeated introductions may be necessary in some cases to achieve complete removal. TM stocking in Carlson Lake was able to meet our fishery improvement objectives of abundance reduction without total elimination, likely because (1) low stocking density and the availability of refuge (littoral) habitat for young Brook Trout prevented complete Brook Trout eradication, and (2) additional introductions of TM were made at critical time periods when Brook Trout abundance was increasing. This is especially true of the relative abundance of Brook Trout <180 mm, which seemed to be the preferred size class for predation during the first few years of TM growth.

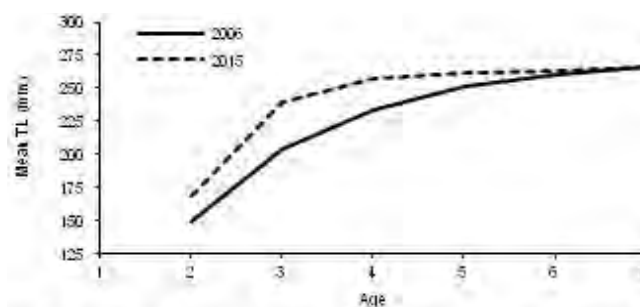


Figure 4. Von Bertalanffy growth curves illustrating differences in Brook Trout growth in Carlson Lake between 2006 (when CPUE was 2.32 fish/hr) and 2015 (when CPUE was 36% lower = 1.44 fish/hr). Values used to construct growth curves were calculated in FAMS (Slipke and Maceina 2014) (2006: $t_0 = 0.652$, $k = 0.588$, and $L_\infty = 272.132$) and (2015: $t_0 = 1.255$, $k = 1.366$, and $L_\infty = 270.62$).

The methods we used to improve fishery quality at Carlson Lake are likely applicable to other HMLs that contain stunted wild trout populations as well. TM readily consume soft-rayed fusiform prey (Goddard and Redmond 1978), so we believe they would also be effective at reducing abundance of other HML salmonid species. However, individual lake characteristics such as elevation and availability of prey refugia (inlets, outlets, and littoral habitat; Koenig et al. 2015), as well as stocking density, size at the time of stocking, and stocking frequency of TM will likely influence results. It may be a delicate balance to sustain abundance reductions over a long-term period, and continued monitoring of prey responses to predator introductions will help develop prescriptive treatments in these cases to maximize sport fish

Table 2. Size structure (mean TL mm and mean relative weight W_r) of each stocked group of tiger muskellunge (2002, 2006, and 2013) sampled at Carlson Lake during the study period, (2002 - 2017).

Year	Stocked in 2002			Stocked in 2006			Stocked in 2013		
	n	Mean TL (range) (mm)	Mean W_r (range)	n	Mean TL (range) (mm)	Mean W_r (range)	n	Mean TL (range) (mm)	Mean W_r (range)
2002	321	300 (160-400)	--						
2003	15	531 (460-580)	112 (96-135)						
2005	8	733 (600-813)	--						
2006	4	735 (710-770)	--						
2009	--	--	--	40 ¹	300 (155-395)	--			
2013	1	1,067	--	3	770 (770-770)	--			
2016	--	--	--	4	899 (863-915)	--			
2017	--	--	--	1	910	100	70 ¹	333 (290-380)	--
							5	708 (647-770)	112 (100-122)
							7	795 (750-865)	115 (105-121)

¹tiger muskellunge measured at the time of stocking

potential. Continued monitoring at Carlson Lake showed us that stocking TM every 4-5 years at 10 – 20 fish/ha can continually improve Brook Trout size structure over a long-term period.

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Joe Chapman and the staff at Hagerman Hatchery raised tiger muskellunge for this project and thus were critical to its success. Arnie Brimmer, Kimberly Murphy, Tom Curet, Bob Esselman, Jon Flinders, and Marsha White collected large portions of this data throughout the duration of the project. Thanks also to Kevin Meyer for providing edits. Funding for this work was provided by anglers and boaters through their purchase of Idaho fishing licenses, tags, and permits, and from federal excise taxes on fishing equipment and boat fuel through the Sport Fish Restoration Program.

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Production and Evaluation of YY-Male Brook Trout to Eradicate Nonnative Wild Brook Trout Populations

Patrick Kennedy¹, Daniel J. Schill¹, Kevin A. Meyer¹, Matthew R. Campbell²,
Ninh Vu², and Michael J. Hansen³

¹Idaho Department of Fish and Game, 1414 East Locust Lane, Nampa, Idaho 83686 (208-465-8404),
(pat.kennedy@idfg.idaho.gov), (dan.schill@idfg.idaho.gov), (kevin.meyer@idfg.idaho.gov)

²Idaho Department of Fish and Game, 1800 Trout Rd., Eagle, ID 83616,
(matt.campbell@idfg.idaho.gov), (ninh.vu@idfg.idaho.gov)

³U.S. Geological Survey, Great Lakes Science Center, Hammond Bay Biological Station, 11188 Ray Road,
Millersburg, Michigan 49759, USA, (michaelhansen@usgs.gov)

Abstract—Nonnative Brook Trout *Salvelinus fontinalis* were introduced throughout western North America in the early 1900s, resulting in widespread self-sustaining populations that are difficult to eradicate and often threaten native salmonid populations. A novel approach for their eradication involves use of YY male (M_{YY}) Brook Trout (created in the hatchery by feminizing XY males and crossing them with normal XY males). If M_{YY} Brook Trout survive after stocking, and reproduce successfully with wild females, in theory this could eventually drive the sex ratio of the wild population to 100% males, at which point the population would not be able to reproduce and would be eradicated. This study represents the first successful development of a F_{YY} and M_{YY} salmonid broodstock, which was produced in four years at relatively low cost. Field trials demonstrated that stocked hatchery M_{YY} Brook Trout survived and produced viable M_{YY} offspring in streams, although reproductive fitness appeared to have been lower than their wild conspecifics. Even if reduced fitness is the norm in both streams and alpine lakes, our population simulations suggest that eradication can be achieved in reasonable time periods under some M_{YY} stocking scenarios, especially when wild Brook Trout are simultaneously suppressed in the population.

Introduction

Brook Trout *Salvelinus fontinalis* have been artificially introduced in many lakes and streams outside their native range and continue to colonize new habitats in western North America. Nonnative Brook Trout populations have negatively impacted native salmonid populations through hybridization, competition, and predation. Thus, fisheries managers have worked to eliminate some exotic Brook Trout populations most commonly using piscicides and electrofishing (Gresswell 1991; Shepard et al. 2014). However, the former negatively impacts non-target aquatic fauna and the latter has resulted in mixed success. Complications with both of these methods points to a need for alternate or companion methods for eradicating nonnative fish.

Gutierrez and Teem (2006) suggested a novel approach that would shift the sex ratio of a wild fish population toward males by annually introducing

hatchery produced male fish with a YY genotype (known as “supermales” but herein referred to as M_{YY} fish). To create a M_{YY} brood stock, normal M_{XY} males are feminized by exposing them to estrogen. The resulting F_{XY} fish are crossed with normal M_{XY} males and one-quarter of the subsequent progeny are M_{YY} (Teem and Gutierrez 2010). By exposing half of the M_{YY} fish to estrogen, an M_{YY} and F_{YY} brood stock can be created, and all offspring are M_{YY} . Large numbers of these M_{YY} offspring can then be reared and stocked into wild fish populations to drive the sex ratio of the wild population to 100% males, theoretically resulting in population eradication (Gutierrez and Teem 2006). Such a stocking program has not been tested in the wild to eradicate a nonnative fish species, though monosex culture is commonly used in commercial hatcheries for artificial fish production (Schill et al. 2016).

Sex ratios in Brook Trout populations would only change under such a stocking program if the M_{YY} fish survive adequately and successfully reproduce after

stocking. In order to shift the sex ratio significantly toward eradication, stocking and suppression will have to be conducted annually for many years. Hatchery trout encounter many challenges upon their liberation into the wild, often leading to very low survival after stocking, especially in streams (Miller 1952). Competition with wild resident fish has been identified as a primary factor contributing to the low survival of hatchery trout in streams (Miller 1958), suggesting that suppression of wild fish prior to stocking hatchery M_{YY} fish could improve survival of the hatchery fish.

From 2008 to 2016 the Idaho Department of Fish and Game (IDFG) conducted a series of studies to (1) develop a M_{YY} hatchery Brook Trout broodstock, (2) evaluate post-release survival and reproductive success of M_{YY} Brook Trout, and (3) model how long it might take to eradicate undesirable Brook Trout populations in both streams and alpine lakes. Herein we briefly summarize the findings of these three studies.

Methods

M_{YY} Brook Trout Broodstock Development

For further details on M_{YY} Brook Trout broodstock development, see Schill et al. (2016). Phase 1 involved the creation of feminized, genetically male fish (F_{XY}). Normal fertilized Brook Trout eggs were hatched (winter 2008/2009) under typical hatchery operations. At swim-up, fry were split into two groups. One group was fed for 60 d with commercially-produced salmonid starter diets treated via spraying with 17β -estradiol (hereafter estradiol) at a concentration of 20 mg steroid per kilogram diet; the other group was fed the same food without estradiol. A subsample of male fish from the untreated group was later used as standard M_{XY} breeders at the beginning of Phase 2 (Figure 1). Fish within each crossing were reared separately to ensure that no siblings were bred together in subsequent generations. At 309 d post-hatch (mean length of about 130 mm TL), all fish were PIT tagged in the body cavity for individual identification, and fin clipped for genetic sex identification.

Concurrently, a suite of microsatellite markers were screened for their ability to genetically determine Brook Trout sex. These efforts proved successful, and in the fall of 2010, a suite of markers were used to individually identify F_{XY} and F_{XX} fish; genotypic F_{XX} females were then culled. The remaining fish in the treated and untreated groups were then examined physically to determine maturation (Figure 1).

Although phenotypic sex could be determined in most fish via secondary sex characteristics, all genetic F_{XY} fish in the treatment groups were examined via a hand-held ultrasound system to identify egg producing fish, which were held separately until mature. All identified F_{XY} fish were either spawned in Phase 2 or were subsequently necropsied to evaluate the proportion successfully feminized.

Phase 2 of this effort involved the development of M_{YY} fish. In November 2010, F_{XY} fish were crossed with standard M_{XY} males. Developing eggs were separated by spawn-pairings and reared as in Phase 1. Standard genotypic crosses yielded 50% M_{XY} , 25% F_{XX} , and 25% M_{YY} fish. These progeny were split into treated and untreated family groups. The treatment groups were again fed estradiol-treated starter feed (as described above), whereas untreated groups were fed an identical untreated diet. Fish were reared by individual spawn-pairings until they were large enough for tagging (fall of 2011).

Genetic markers were again used to identify XX, XY, and YY fish in the raceway. When these fish reached maturity (October 2012), maturing F_{YY} and M_{YY} fish in the raceway were identified by combining both genetic marker information and observed phenotypic secondary sex characteristics (Figure 1). Once genotypes and phenotypes were available, all F_{XX} , F_{XY} , and M_{XY} fish present in the raceway were culled, leaving only F_{YY} and M_{YY} fish (i.e., the M_{YY} broodstock). In Phase 3, these fish were spawned to produce M_{YY} fish for subsequent pilot field trials (see below) and to renew the M_{YY} broodstock.

Field Trials of Survival and Successful Spawning

For further details on field evaluations, see Kennedy et al. (2016). Two treatment levels were implemented for this evaluation: (1) suppression of the wild Brook Trout via electrofishing removal prior to M_{YY} stocking to reduce potential competition (Wildhorse and Bear creeks); and, (2) no suppression with M_{YY} stocking only (Iron Bog and Cherry creeks). The study streams were selected for known Brook Trout populations and relatively narrow stream width so we could efficiently conduct backpack electrofishing removals in this pilot study. All M_{YY} fish were adipose fin clipped prior to stocking so they could be differentiated from wild Brook Trout in the stream.

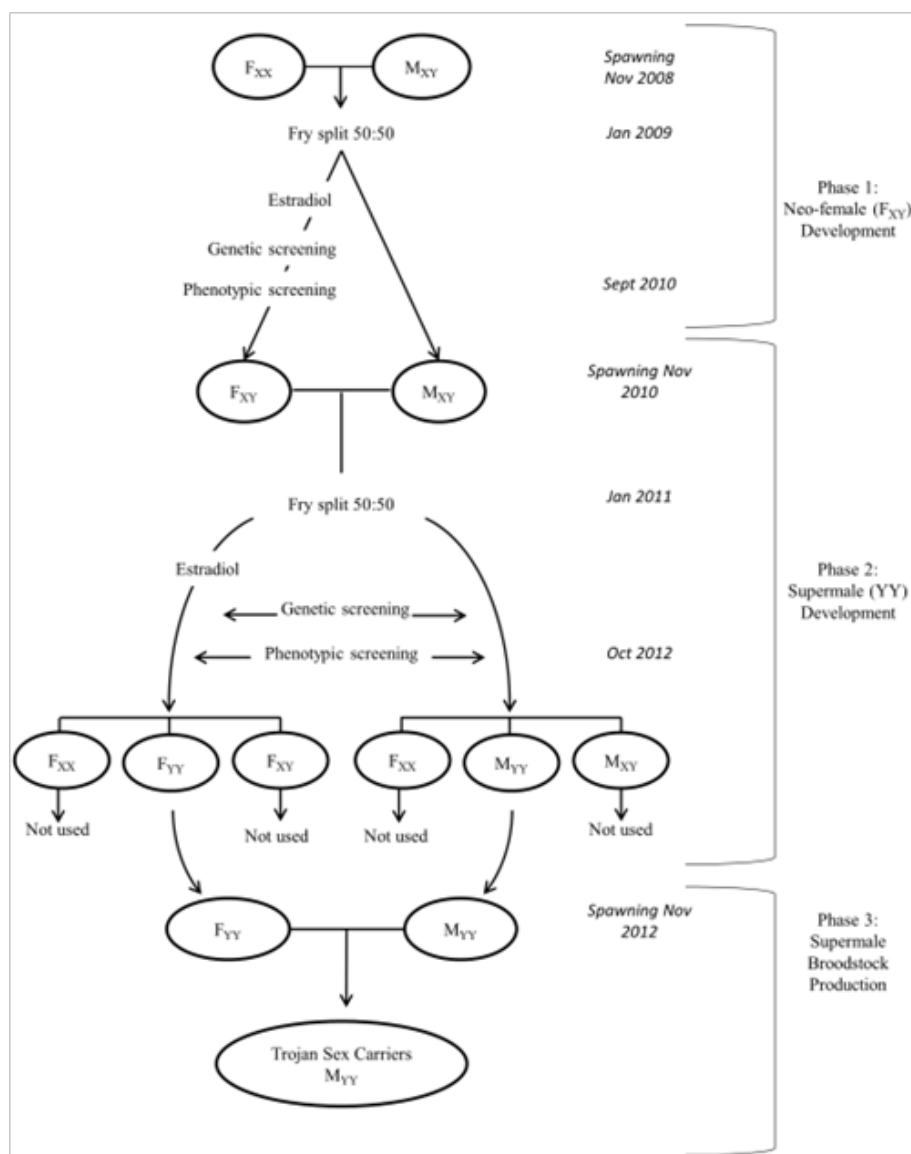


Figure 1. Schematic outlining general method of M_{YY} Brook Trout production, 2008-2012.

In early June of 2014, At Bear and Wildhorse creeks, all wild Brook Trout captured via electrofishing were euthanized and measured to the nearest mm. In late June of 2014, approximately 500 M_{YY} Brook Trout were evenly dispersed throughout the study reach of each study stream (Table 1).

In October of 2014, mark-recapture electrofishing was conducted in each study reach to estimate the abundance of adult wild (≥ 100 mm) and M_{YY} Brook Trout. Single-pass electrofishing was also conducted 300 m above and below each study reach to estimate post-stocking emigration of M_{YY} Brook Trout out of the study reaches. Emigration rates were estimated

by dividing the proportion of captured emigrants from each study reach by the capture efficiency for M_{YY} Brook Trout in the mark-recapture surveys. Unadjusted survival of M_{YY} Brook Trout was estimated by dividing the abundance of M_{YY} Brook Trout estimated within the study reach in October by the number originally stocked. Apparent survival was calculated by subtracting the emigration rate from the unadjusted survival estimate.

Angler exploitation of M_{YY} trout after stocking was estimated following the methods of Meyer and Schill (2014). Approximately 10% were tagged prior to stocking using T-bar anchor tags. Anglers could

report tags through the IDFG phone system, website, regional offices, or by mail. We assumed that anglers reported 41% of these non-reward tags (K. Meyer, unpublished data). We calculated angler exploitation through four months after stocking.

Tissue samples were collected from M_{YY} Brook Trout at the hatchery and from wild Brook Trout at each stream. To determine if M_{YY} Brook Trout successfully reproduced in the wild, approximately 100 tissue samples were collected and genotyped from Brook Trout fry (< 90 mm) the following year (in Fall of 2015) at each study stream. All samples were screened with 240 single nucleotide polymorphic (SNP) loci. Genotyping followed protocols developed by Campbell et al. (2015). Putative first generation (F_1) offspring from M_{YY} Brook Trout were identified using program STRUCTURE version 2.3.3 (Pritchard et al. 2000) to estimate individual membership coefficients (Q). A total of 50,000 Markov Chain Monte Carlo samples were drawn after discarding the first 10,000 iterations. We created simulated F_1 offspring between known M_{YY} Brook Trout and wild individuals using functions in program Excel. For each study population, 10 simulated offspring genotypes were created by crossing 5 M_{YY} “parents” with 5 wild “parents” from each study stream. The admixture proportions observed in the simulated F_1 offspring were used as criteria to assign juveniles as F_1 offspring from M_{YY} and wild females.

Simulated Time to Eradicate Brook Trout Populations

For further details regarding eradication modeling, see Schill et al. (2017). Briefly, we constructed an age-structured stochastic model to simulate effects of a range of fishing mortality (imposed via manual suppression) and M_{YY} stocking rates on long-term viability of hypothetical wild Brook Trout populations. For stream evaluations, we parameterized the model to mimic the Brook Trout population in Hunt Creek, Michigan, during 1949–1962 (McFadden et al. 1967), because population demographics were similar to introduced Brook Trout in western North America (e.g., Meyer et al. 2006). The population growth rate (R) in each year was treated as a function of year-specific total abundance and an assumed carrying capacity (K) of 10,000 total fish of all ages, based on a reasonable density of 1,000 total Brook Trout per stream km (Meyer et al. 2006) and a hypothetical

stream length of 10 km. The maximum population growth rate (R_{max}) for simulated populations was set at the highest net reproduction rate estimated for the Brook Trout population in Hunt Creek.

Simulated fishery management actions included a range of suppression rates (via electrofishing removals) and M_{YY} stocking rates, both conducted annually. In practice, fish stocked in streams would be adipose fin-clipped to distinguish them from wild fish during electrofishing, so they could be released. Wild fish suppression was simulated for three rates (25, 50, and 75% of the population), in conjunction with relative age-specific selectivity to backpack electrofishing gear derived from typical recapture rates of marked fish in Idaho streams. Stocking of M_{YY} fingerlings was incorporated into the models at three proportions (10, 25, and 50%) of the expected number of age-0 Brook Trout (6,640 fish) present at the simulated K and average age-specific survival rates of the simulated population. Fitness (survival and reproductive success) was initially assumed to be the same for stocked M_{YY} Brook Trout as for their wild counterparts. To evaluate less than optimal fitness of stocked M_{YY} fish relative to wild fish, we also ran simulations assuming that stocked fish were only 20% as fit as wild fish.

Modeling of alpine lake populations was the same as for streams except that parameter values were set to mimic abundance and survival of Brook Trout in Idaho alpine lakes (M. Koenig, IDFG, unpublished data). Population K was set at 3,500 fish of all ages. Suppression and M_{YY} stocking levels were the same as for streams, but suppression would require use of lethal overnight gill netting in alpine lakes, thus stocked M_{YY} fish were subjected to the same suppression in years after stocking as wild fish.

For each water type and combination of suppression and stocking rates, 1,000 iterations of the model were run for a 50-year period. Time to eradication for each combination of removal and stocking rate was represented as the year that total abundance of all age groups declined to zero for all simulations.

We also modeled years to eradication in streams and alpine lakes across a wider range of stocking rates than established above. For these simulations, we modeled suppression rates of 0, 25, and 50%, assumed that M_{YY} fish were as fit as wild fish, and varied stocking rates from 0 to 100% in 10% increments.

Assuming that M_{YY} fish are only 20% as fit as wild fish, results from these simulations equated to stocking rates of 0 to 500% in the poor fitness scenario.

Results

YY Brook Trout Broodstock Development

In Phase 1, sex reversal of genetically male (M_{XY}) Brook Trout to F_{XY} females did not prove difficult. Necropsies performed on putative F_{XY} females genetically identified from the estradiol treatment group (but not needed for YY broodstock production) indicated a near 100% success in full feminization to F_{XY} females, as only one individual intersex fish was observed. Thus for all treated XY Brook Trout reared to maturity, 223 of 224 fish or 99.6% had fully-formed, functional ovaries and were considered phenotypic females.

During Phase 2, use of both genetic sex markers and phenotypic screening of pre-spawners, 51 maturing feminized F_{YY} fish and 49 maturing M_{YY} fish were preliminarily identified in October 2012. In Phase 3, of the 19 $F_{YY} \times M_{YY}$ crosses made, six failed to produce viable progeny while the remaining crosses yielded more than enough viable M_{YY} green eggs for future YY broodstock and for the field experiment described below.

Field Trials of Survival and Successful Spawning

In June of 2014, a total of 2,010 MYY Brook Trout catchables produced in Phase 3 above were stocked in the study streams (Table 1). Prior to stocking, we removed 1,026 wild Brook Trout from Bear Creek

and 210 from Wildhorse Creek. By October, across all streams, we estimated 226 MYY fish and 8,266 wild Brook Trout (>100 mm) remained in the study reaches; therefore, MYY fish comprised 2.7% of the spawning Brook Trout population (wild and hatchery). Capture efficiencies averaged 57% (range 43-74%) for wild Brook Trout and 83% (range 75-100%) for MYY fish. Considering our abundance estimates and assuming no mortality of wild fish from June to October, MYY fish were presumably stocked at an average of 27% (range = 15-40%) of the wild Brook Trout population across all study streams, and the removals would have constituted a 24% suppression of wild fish in Bear Creek and a 9% suppression in Wildhorse Creek.

Emigration of M_{YY} Brook Trout out of the study reach and into adjacent stream reaches averaged 1.2% (Table 1). Angler exploitation was nil in Bear and Cherry creeks and averaged 24.8% in Wildhorse and Iron Bog creeks. Survival averaged 16.1% at suppression streams and 9.1% at non-suppression streams. In 2015, 14 individuals with genotypes indicating $F_1 M_{YY}$ offspring were detected in all study streams combined, and M_{YY} offspring were detected in each of the study streams (Table 2). All 14 individuals identified as F_1 offspring were genetic XY males.

Simulated Time to Eradicate Brook Trout Populations

Simulations of the time required to eradicate Brook Trout populations using M_{YY} stocking revealed that if stocked M_{YY} fish survive and reproduce as well as wild males in streams, Brook Trout could be eradicated using several combinations of M_{YY} stocking and electrofishing suppression. For example, time to

Table 1. Estimates (Est.) of and 90% confidence intervals (CI) for the abundance of wild Brook Trout (>100mm) and the abundance, exploitation, emigration, and survival of M_{YY} Brook Trout stocked in four study streams in central Idaho.

		Wild Brook Trout			M _{YY} Brook Trout									
		October			Stocked	October			Angler		Emigration		Apparent	
		abundance				in June	abundance			exploitation (%)		(%)		survival (%)
Creek	Treatment	Est.	CI	<i>p</i> (%)			Est.	CI	<i>p</i> (%)	Est.	CI	Est.	CI	Est.
Bear	Suppressed	3,254	185	47	492	103	18	75	0.0	0.0	2.4	1.0	23.4	3.6
Wildhorse	Suppressed	2,034	114	43	506	43	3	83	27.5	16.0	0.2	0.3	8.7	0.6
Cherry	Not suppressed	1,682	67	62	500	44	3	75	0.0	0.0	0.5	0.5	9.4	0.6
Iron Bog	Not suppressed	1,296	60	74	512	36	0	100	22.0	14.4	1.8	1.0	8.8	0.0

Table 2. Sample group along with treatment level (suppression or non-suppression) and stream name. Sample size is shown for each sample group, along with minimum, maximum, and average proportional membership observed.

Creek	Treatment level	Sample group	Sample size	Proportional membership		
				Min	Max	Avg.
Bear	Suppressed	M _{yy} offspring	3	0.456	0.554	0.509
Wildhorse	Suppressed	M _{yy} offspring	5	0.383	0.605	0.490
Cherry	Not suppressed	M _{yy} offspring	4	0.417	0.552	0.500
Iron Bog	Not suppressed	M _{yy} offspring	2	0.405	0.644	0.525

Table 3. Predicted years to eradication and 95% lower (LCI) and upper (UCI) confidence intervals for Brook Trout in hypothetical streams and alpine lakes in Idaho subjected to a range of selective electrofishing (streams) and non-selective gill-netting (lakes) suppression rates and M_{yy} stocking rates. Predictions assumed M_{yy} fitness (survival and reproductive success) was equivalent to wild males (good survival) and 20% of wild males (poor survival).

Suppression rate	Stocking rate	Streams						Alpine lakes					
		Good survival			Poor survival			Good survival			Poor survival		
		Years	LCI	UCI	Years	LCI	UCI	Years	LCI	UCI	Years	LCI	UCI
0%	10%	>50	>50	>50	>50	>50	>50	>50	1	>50	>50	1	>50
	25%	>50	9	>50	>50	>50	>50	23	1	25	>50	1	>50
	50%	12	3	12	>50	>50	>50	8	1	8	>50	1	>50
25%	10%	>50	13	>50	>50	>50	>50	>50	1	>50	>50	1	>50
	25%	13	4	14	>50	>50	>50	14	1	15	>50	1	>50
	50%	6	3	7	>50	14	>50	8	1	8	>50	1	>50
50%	10%	13	5	14	>50	15	>50	20	1	23	>50	1	>50
	25%	6	3	7	26	8	28	10	1	10	>50	1	>50
	50%	4	2	4	12	5	15	7	1	8	18	1	21
75%	10%	6	4	6	10	6	11	11	1	13	25	1	30
	25%	4	2	4	7	4	8	7	1	8	16	1	19
	50%	4	2	4	6	4	6	6	1	7	11	1	12

eradication was 12–13 years for 10% stocking and 50% electrofishing suppression, 25% stocking and 25% suppression, or 50% stocking and 0% suppression (Figure 2, Table 3). Similarly, time to eradication was as little as 6 years for 10% stocking and 75% electrofishing suppression, 25% stocking and 50% suppression, or 50% stocking and 25% suppression. If stocked M_{yy} fish are only one-fifth as fit as wild males in streams, Brook Trout could be eradicated only by using high rates of stocking, with or without concurrent electrofishing suppression.

In alpine lakes, if M_{yy} fish survive and reproduce as well as wild males, Brook Trout could be eradicated using several combinations of M_{yy} stocking and gill

net suppression, but time to eradication was longer than in streams. For example, time to eradication was 8–11 years at 10% stocking and 75% gill netting suppression, 25% stocking and 50% suppression, and 50% stocking and 25% suppression (Figure 2, Table 3). Population eradication was achievable in 10 years or less only at a stocking rate of 50% or greater (regardless of the suppression rate), or a stocking rate of 25% and a suppression rate of 50% or greater. If M_{yy} fish are one-fifth as fit as wild males in alpine lakes, Brook Trout could only be eradicated by using very high rates of stocking and gill net suppression. For example, Brook Trout were only eradicated by a suppression rate of 75%, regardless of stocking

rate, or when both stocking rate and suppression rate were 50% or greater. Eradication was not achievable within 10 years in alpine lakes at any of the initial combinations of suppression and stocking rates when we assumed that stocked M_{YY} fish were 80% less fit than wild fish.

Across a broader range of potential stocking rates, suppression rate influenced years to eradication for hypothetical Brook Trout populations more dramatically in streams than in alpine lakes. For

example, reducing suppression in streams from 50% to 25% would require more than a doubling of the stocking rate to maintain a 10-year eradication time frame, whereas in alpine lakes, the same reduction in suppression would require only a 40% increase in stocking rate to maintain a 10-year eradication time frame (Figure 3). Assuming that M_{YY} fish are as fit as wild males, any stocking rate greater than 49% in alpine lakes or 60% in streams achieved eradication in 10 years or less, regardless of the suppression rate.

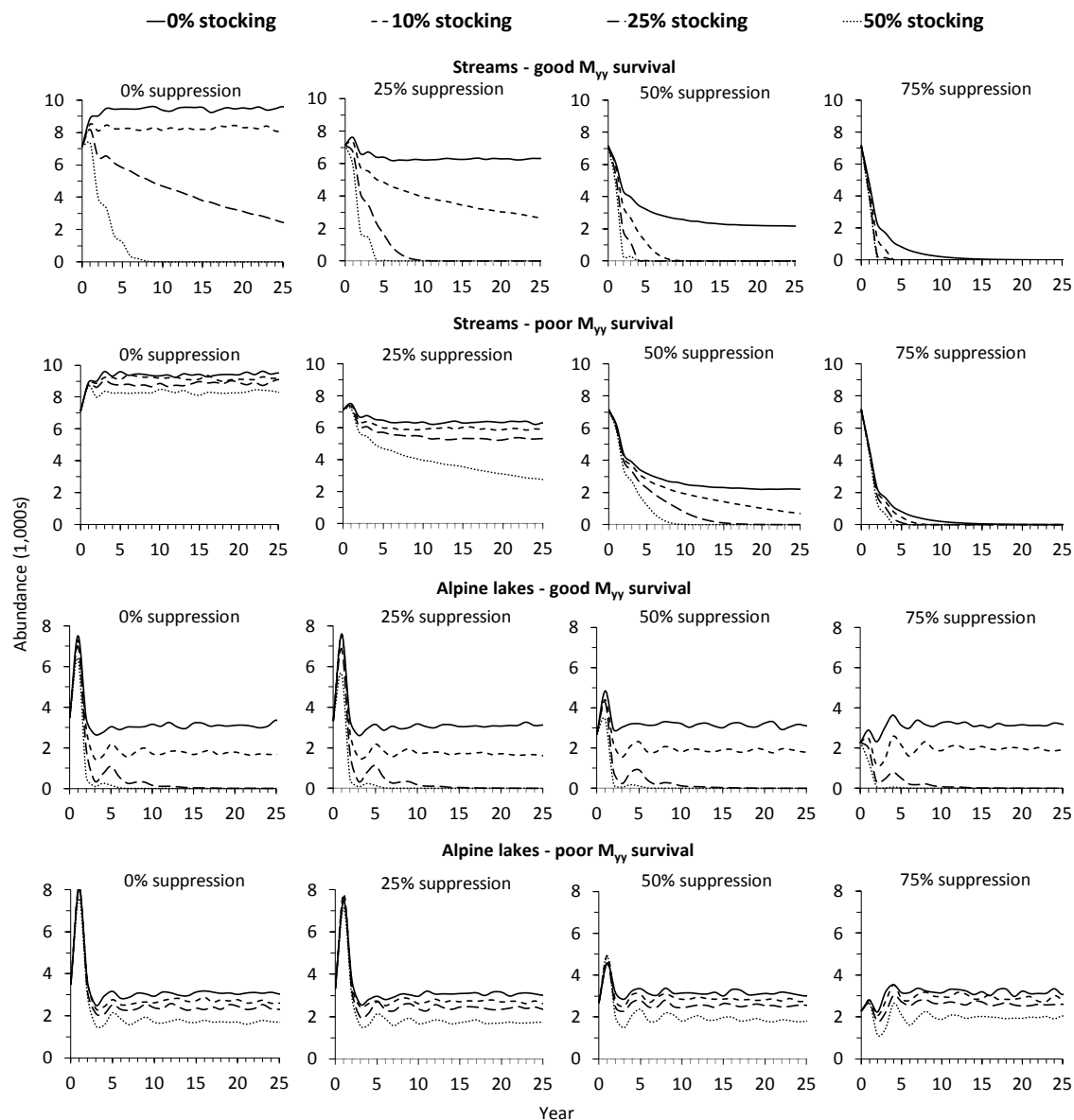


Figure 2. Simulated abundance of Brook Trout in hypothetical streams and alpine lakes in Idaho subjected to a range of suppression and MYY stocking rates, assuming MYY survival was equivalent to wild males (good survival) and 20% of wild males (poor survival).

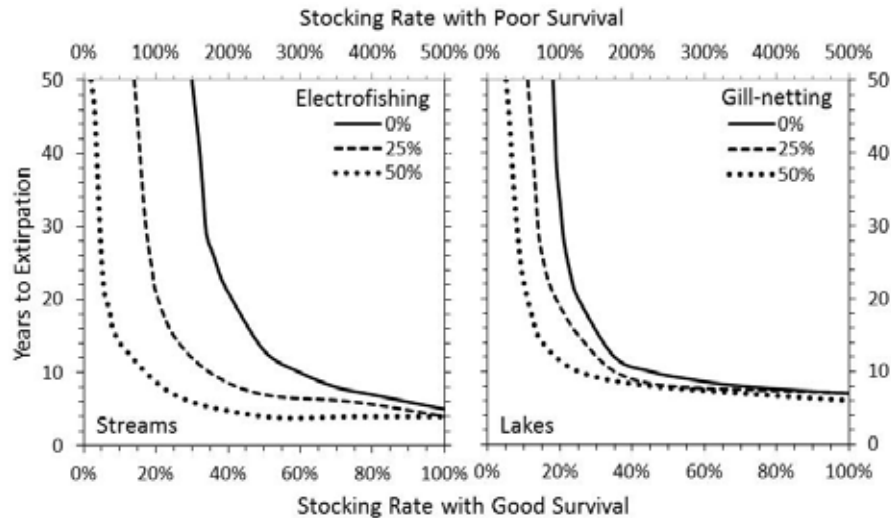


Figure 3. Predicted years to eradication of Brook Trout in hypothetical streams and alpine lakes in Idaho subjected to a range of suppression and M_{YY} stocking rates, assuming M_{YY} fitness (survival and reproductive success) was equivalent to wild male fitness (good survival) and 20% of wild male fitness (poor survival).

Discussion

YY Brook Trout Broodstock Development

Although the process we used to produce a M_{YY} Brook Trout broodstock sounds complex, monosex production of numerous species has been accomplished for decades in commercial aquaculture. In our case, nearly all of the elapsed time from project initiation (Fall of 2008) to successful F_{YY} and M_{YY} broodstock spawning (Fall of 2012) was to allow fish to mature between production phases. Labor was primarily focused in 2- to 3-d spawning periods at the end of each production phase and for PIT-tagging and fin clipping. Total costs to develop the broodstock, including genetic testing, feed, and labor, were probably less than US \$30,000, in part because a sex-linked genetic sequence had presumably already been identified and thus the costs of developing a working sex marker were minimal. The availability of such sex markers has been identified as a crucial step in developing a M_{YY} program (Cotton and Wedekind 2007).

Field Trials of Survival and Successful Spawning

Across all streams combined, the M_{YY} Brook Trout we stocked comprised an estimated 2.7% of

all spawning adults in 2014 and they produced an estimated 3.7% of the progeny that year. While these findings demonstrate that hatchery M_{YY} Brook Trout can successfully compete reproductively with wild male conspecifics in streams, these numbers do not indicate that M_{YY} fish were more prolific, since only a portion of the wild Brook Trout were mature males. Considering the estimated sex ratios at Bear and Wildhorse creeks (data not shown), and assuming that male Brook Trout in Idaho streams become sexually mature at about 150 mm, and limiting abundance estimates to fish greater than 150 mm for these two streams, we estimate that M_{YY} fish comprised about 10% of all spawning male Brook Trout at Bear and Wildhorse creeks combined, and produced 4% of the progeny, suggesting that while M_{YY} Brook Trout are reproductively capable, they are not as fit as their wild conspecifics.

Apparent survival was low for hatchery M_{YY} Brook Trout but was similar to results from other studies of catchable-sized hatchery trout stocked in streams (e.g., Miller 1952). Hatchery trout survival can be increased by reducing wild fish abundance (Miller 1958), and our study showed some evidence of improved survival of M_{YY} fish in suppression streams. Post-release survival was reduced only slightly by emigration and exploitation. When considering an M_{YY}

stocking program to eradicate undesirable fish, one question worth evaluating is which size of hatchery fish to stock. Rearing fish to fry and fingerling size is much less expensive and requires much less hatchery rearing space, but post-release survival is typically much lower, and the fish must survive at least 1-2 years to reach maturity before they can spawn with wild fish.

Simulated Time to Eradicate Brook Trout Populations

Our simulations suggest that stocking M_{YY} fish could eradicate wild nonnative fish populations in reasonable timeframes, whereas previous studies have generally suggested that many decades would be required to eradicate populations (Gutierrez and Teem 2006; Teem and Gutierrez 2010). Such slow responses are unlikely to be acceptable to fishery managers or the public, and may partly explain why management interest in a M_{YY} approach has been limited to date. Responses predicted by previous studies were slower than our predictions for several reasons. First, previous studies have generally modeled low (4–7%) stocking rates of reproductively competent adults, whereas we modeled fingerling stocking rates as high as 100% of the existing wild fish. Such stocking rates are certainly feasible for many undesirable Brook Trout populations in western North America, and our results demonstrate the importance of stocking at higher rates to quickly eradicate populations. Second, Brook Trout have a shorter generation time than most other nonnative fish species, thereby making them more vulnerable to sex-skewing eradication methods. Third, no prior M_{YY} simulation study has included concurrent manual suppression, which our results suggest can greatly speed the eradication process.

Our simulations suggest longer time to eradication and higher stocking rates would be needed to eradicate Brook Trout in alpine lakes compared to streams, in part because Brook Trout mature later and live longer in lakes than in streams, which slows the demographic input of successful M_{YY} spawning in the wild population. Also, lethal overnight gill netting removes some M_{YY} fish from lakes that are not killed with selective electrofishing removals in streams. A likely reason for the relative insensitivity of our alpine lake results to suppression above 40% was use of non-selective gill nets, because stocked M_{YY} fish would also be killed at the same high rates as wild males.

Although fitness of stocked fingerling M_{YY} Brook Trout may be lower than that of wild fish, such results are not a certainty in all instances. For example, suppression of wild Brook Trout may have increased survival of stocked M_{YY} catchables in our study streams. In alpine lakes, stocked M_{YY} fingerlings may benefit from increased age-0 survival and an associated recruitment pulse that was consistently found in California alpine lakes following sustained wild Brook Trout removal via gill netting (Hall 1991).

Conclusion

To our knowledge, this effort represents the first successful construction of a F_{YY} and M_{YY} salmonid broodstock, which IDFG staff produced in 4 years and with minimal cost. Our field trials demonstrated that stocked hatchery M_{YY} Brook Trout survived and produced viable M_{YY} offspring in streams, although fitness appeared to have been lower than their wild conspecifics. Even if reduced fitness is to be expected in both streams and alpine lakes, our population simulations suggest that eradication can be achieved in reasonable time periods under some stocking and wild suppression scenarios. Based on these findings, IDFG initiated (in 2016) a broad-scale evaluation (eight alpine lakes and eight streams in Idaho) to determine the potential for an M_{YY} stocking program to eradicate undesirable wild Brook Trout populations; both fingerling and catchable-sized M_{YY} Brook Trout are being evaluated.

Public acceptance will be an important issue with such a program, although for several reasons we believe this should not pose a major challenge. First, no fish destined for human consumption have been treated with hormones. Second, the method is specific to the target exotic species, not native species, so there is little to no possibility of direct ecological collateral damage. Perhaps most importantly in regard to public acceptance, a M_{YY} fish is not a genetically modified food organism (GMO) since no new genetic material is infused into released fish. For this reason, we suggest that an M_{YY} program is the least likely of various “genetic” approaches for exotic fish suppression to generate public controversy (Thresher et al. 2013). Finally, it is also worth noting that the amount of hormone released into the aquatic environment for development of the existing YY broodstock is inconsequential (Schill et al. 2016).

This work may represent a major advancement toward eradicating undesirable nonnative Brook Trout populations where they threaten native species or provide inconsequential fisheries for anglers. While our field trials demonstrated successful M_{YY} reproduction, further evaluations are needed to determine if this program is effective at actually shifting the sex ratio to the extent that a wild population of Brook Trout can be eradicated. If so, stocking M_{YY} fish could become an effective and economic method of eliminating nonnative Brook Trout populations across western North America.

In an effort to present a brief synthesis of IDFG's YY Male Brook Trout work, this written document briefly summarizes three separate aspects of the program. The reader is referred to more detailed written summaries for more specifics (Kennedy et al. 2016; Schill et al. 2016; Schill 2017).

Acknowledgments

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Session 7

Stream Habitat Management: Traditional and New Approaches





Who is Doing What to Trout: Insights from the National Fish Habitat Assessment

Gary E. Whelan¹, Wesley M. Daniel², Emily Dean², Dana Infante², Kyle Herreman², and Arthur Cooper²

¹Michigan Department of Natural Resources, Fisheries Division, P.O. Box 30446, Lansing, Michigan 48909, USA

²Department of Fisheries and Wildlife, Michigan State University, 1405 South Harrison Road, Suite 318, East Lansing, MI 48823, USA

Abstract—As part of the science focus of the National Fish Habitat Partnership Board (NFHP), a national assessment of fish habitat is conducted every 5 years and the 2015 assessment is the second such assessment. The river assessment component uses a landscape-scale approach and calculates risk of fish habitat degradation by a regionally developed fish-based approach that uses fish abundance data and relates these to human development factors using a regression tree analytical approach for the lower 48 states. This study used abundance data from the national assessment for Cutthroat Trout *Oncorhynchus clarkii*, Rainbow Trout *Oncorhynchus mykiss*, Brown Trout *Salmo trutta*, and Brook Trout *Salvelinus fontinalis* to examine landscape-scale human development effects and development factors that are the most pervasive and severe disturbances. Brown Trout showed the least number of landscape impairments and as expected, are likely the most tolerant to anthropogenic landscape factors. Similarly, Cutthroat Trout appeared to be the least tolerant of this group of salmonids. The network scale was much more sensitive than the local spatial scale. Key anthropogenic factors negatively affecting these trout species were low-intensity urban land use, road length density, human population density, road density, pasture, sediment yield, crop land, and impervious surface.

Introduction

Aquatic habitat degradation is one of the primary drivers of freshwater fish loss in North America (Jelks et al. 2008). Land-use change from natural lands to urbanized (5% currently, NLCD 2011) and/or agricultural lands (23% currently, NLCD 2011) degrade stream fish habitat throughout the United States (Richards et al. 1996; Roth et al. 1996; Wang et al. 2003; Allan 2004). These land-use changes fundamentally change the controlling aquatic habitat processes and factors that include hydrology, connectivity, geomorphology, material recruitment and transport, water quality, and energy flow, in turn altering the fish communities of the affected watersheds.

Of the freshwater fishes at risk from land-use change, salmonids are a particularly sensitive group to anthropogenic changes with many stream resident trout species (Salmoninae) identified as imperiled (Jelks et al. 2008). In the U.S., there are currently 29 native trout species and subspecies and one nonnative naturalized trout species (Williams et al. 2015). Of the native trout, six are listed as threatened or endangered and over half (13 species) inhabit less than 25% of their native ranges (Williams et al. 2015).

Evaluating waters using a landscape approach links instream habitat characteristics and biological processes with the characteristics of the surrounding landscape (Allan 2004) and is a powerful tool to understand the ecology of stream fishes along with their potential stressors (Schlosser 1991; Fausch et al. 2002). Multiple studies have examined the relationship between stream resident trout and U.S. landscapes (May et al. 1999; Wang et al. 2003; Vondracek et al. 2005; Neville et al. 2006; Hudy et al. 2008; Stranko et al. 2008; Berger and Greswell 2009; McKenna and Johnson 2011; Wagner et al. 2013; DeWeber and Wagner 2015). These studies have demonstrated the importance of specific natural landscape characteristics (e.g. drainage area, forest cover, geology) for trout, but generally have not examined the influence of anthropogenic effects and none has been conducted on a national scale.

Since it is known which trout stream habitats in the lower 48 states of the U.S. have been affected by urban, agricultural and other land uses, it is critical to assess the degree to which trout habitat is currently at risk of degradation, to what degree, and where these waters are located to improve abilities to identify and

prioritize trout populations at greatest risk. The goals of this preliminary study are to (1) document the relationship between trout, trout stream habitat, and landscape characteristics; and (2) assess the degree trout and trout stream habitat have been affected by anthropogenic land use at a national scale using data from all of the lower 48 U.S. states. To meet these goals, this study had the following objectives: (1) examine how Cutthroat Trout *Oncorhynchus clarkii*, Rainbow Trout *Oncorhynchus mykiss*, Brown Trout *Salmo trutta*, and Brook Trout *Salvelinus fontinalis* are affected by landscape-scale human development variables; and (2) determine the landscape disturbance factors that are most pervasive and severe for these trout species.

Methods

Study Areas. – This study evaluated trout habitat in the contiguous United States and stratified the analysis to account for regional physical differences by using the ecoregions from the US EPA's National Wadeable Streams Assessment (US EPA 2006). Four ecoregions were used in their entity: the Northern Appalachian region (NAP), Upper Midwest region (UMW), Western Mountain region (WMT), and Xeric region (XER; Figure 1). Two additional regions were created by pooling adjacent ecoregions to ensure adequate sample size: the Southern Coast and Appalachian region (SCA), which combined the Southern Appalachian (SAP) and Coastal Plain (CPL) ecoregions, and the US Plains region (UPL), which combined the Temperate Plains (TPL), Northern Plains (NPL), and Southern Plains (SPL) regions.

Spatial framework. – All data were attributed to an existing National River Spatial Database (NRiSD) - 1:100,000 National Hydrography Dataset Plus Version 1 (NHDPlusV1, NHDPlus, 2008) geospatial framework that includes stream reaches, catchments draining to reaches, and 90-m stream buffers bordering stream reaches (Wang et al. 2016). We attributed stream fish data to individual stream reaches defined by stream confluences (Wang et al. 2011), and attributed landscape data to each of four spatial extents. Local catchments include all land area draining directly into an individual stream reach, and network catchments include all upstream lands throughout the stream network, including local catchments (Wang et al. 2011). Similarly, local buffers include all land area 90 m on either side of stream

reaches, and network buffers include all land area 90 m on either side of the stream network upstream of and including the local buffer.

Fish Data. – Stream fish abundances, including abundances of Cutthroat, Rainbow, Brown, and Brook trout with all subspecies and races assigned to species level, were collected using single pass electrofishing surveys targeting whole fish community assemblage samples from 1990 to 2013 from the 2015 National Fish Habitat Assessment (Crawford et al. 2016). A total of 39,405 stream reaches had fish data with Cutthroat Trout - 2,695 reaches, Rainbow Trout 2,337 reaches, Brown Trout - 2,852 reaches, and Brook Trout - 6,858 reaches. Current known ranges for Cutthroat, Rainbow, Brown and Brook trout were provided by the Nonindigenous Aquatic Species Program at the HUC-8 watershed level (U.S. Geological Survey 2017).

Landscape factors. – For each region, we assembled landscape variables known to be important natural influences on stream fish habitats or stressors to habitat and stream fishes (Esselman et al. 2013). Six natural landscape factors and 26 anthropogenic landscape factors from Crawford et al. (2016) were examined. The anthropogenic landscape factors were summarized in four spatial scales to test relationships between trout relative abundance data and a single landscape factor in different extents because each trout species may have different sensitivities to disturbances at local vs. larger spatial scales.

Analytical approach. – The analytical approach taken for this study follows the steps described in Crawford et al. (2016) attributing available trout abundance and landscape data into NRiSD. We accounted for natural variation in the trout data by testing for spatial autocorrelation between our fish samples sites to eliminate type I errors using the program Spatial Analysis in Macroecology (SAM, Version 4.0; Rangel et al. 2010), and used a boosted regression process to account for the influence of important natural factors on abundance of the trout species (Elith et al. 2008, Esselman et al. 2013). Finally, we used a conservative threshold approach using the TITAN (Baker and King, 2010) and Segmented R Segmented (Muggeo 2013) packages to test all trout species against and identify the important anthropogenic landscape factors then applied developed relationships to create scores in each trout species' known range.

To create the habitat condition index (HCI) for each trout species' streams of the contiguous United States, and the cumulative habitat condition index (CHCI) for all four trout species, thresholds and plateau points were identified from significant response thresholds. This information was then synthesized into a single number by taking the range of the anthropogenic landscape disturbance factor occurring between the threshold and plateau points was divided by three and then assigned each a condition class, reflecting relative condition of habitat. The five condition classes were as follows: very low risk of habitat degradation (5), low risk of habitat degradation (4), moderate risk of habitat degradation (3), high risk of habitat degradation (2), and very high risk of habitat degradation (1). The condition classes assigned to each significant threshold were used to extrapolate scores generated to reaches without sampled trout data within the known ranges of each trout species. The HCI for each trout species was determined using the minimum condition score (i.e. highest risk) for all anthropogenic landscape factors across all spatial extents using the concept that the most limiting score will limit overall habitat condition following the methods of Esselman et al. (2013).

Limiting disturbances were summarized by total stream length and determined for each trout species based on two criteria: (1) pervasive (most common) disturbances forced deviation away from best condition toward a minimum condition score of low, moderate, high or very high risk of habitat degradation; and (2) severe disturbances (a subset of pervasive disturbances) that are associated with stream reaches scored as high or very high risk of habitat degradation.

Results

Cutthroat Trout. – Of the four regions tested in this study, only the WMT and XER regions had significant Cutthroat Trout threshold responses to anthropogenic landscape factors (Figure 1). A total of 26.9% of the threshold responses were significant, and 58.6% of those thresholds occurred in the network catchment spatial extent. Based on the results of the assessment, 56.1% of the Cutthroat Trout's stream kilometers were classified as very low (30.9%) or low (25.3%) risk of habitat degradation (Figure 2), the lowest amount of the four trout species. This species also had the highest combined (30.0%) kilometers of habits classified as very high (21.4%) and high (8.65%) risk of degradation.

There were four significant anthropogenic landscape factors at a local catchment scale, but only three accounted for >10% of kilometers of impaired streams. For pervasive disturbances, road length density (62.3%), percent pasture land-use (17.7%) and percent low-intensity urban land-use (16.6%) were the key factors for this species (Table 1). Those same three landscape factors respectively (38.8%, 36.9%, and 15.8%) accounted for 91.5% of the severe landscape disturbance leading to habitat condition scores of high and very high degradation risk for local catchments. Thirteen significant anthropogenic landscape factors were found at the network scale, with four pervasive factors in >10% of kilometers: road length density (33.3%), percent low-intensity urban land-use (15.7%), anthropogenic sediment yield (15.2%) and road crossing density (14.8%). These factors made up 79.0% of the significant thresholds. When looking at the severe disturbances, three factors accounted for 76.7% of the high and very high risk of degradation scores: road length density (39.3%), percent low-intensity urban land-use (24.3%) and anthropogenic sediment yield (13.1%). Local buffer had one significant pervasive and severe anthropogenic landscape factor, percent pasture land-use as did the network buffer, percent impervious surface. Overall, the most important factors contributing to habitat disturbance risk for Cutthroat Trout were road length and crossing density, pasture land-use, low intensity urban land-use, above background sediment yield, and impervious surfaces.

Rainbow Trout. – Of the six regions used in this study, all but the UPL region had significant Rainbow Trout threshold responses to anthropogenic landscape factors (Figure 1). A total of 14.8% of the threshold responses were significant, and 57.5% of those thresholds occurred in the network catchment spatial extent. Based on the results of the assessment, 61.6% of the Rainbow Trout's kilometers of stream habitat is at very low (27.8%) or low (33.7%) risk of habitat degradation from the tested anthropogenic landscape factors (Figure 2). This species had 25.1% of its river kilometers of stream habitat classified as very high (21.4%) or high (8.65%) risk of degradation.

The most pervasive anthropogenic landscape disturbances to Rainbow Trout at the local catchment scale were road length density (54.9%), human population density (17.6%), percent low intensity urban land use (14.6%), and road crossing density

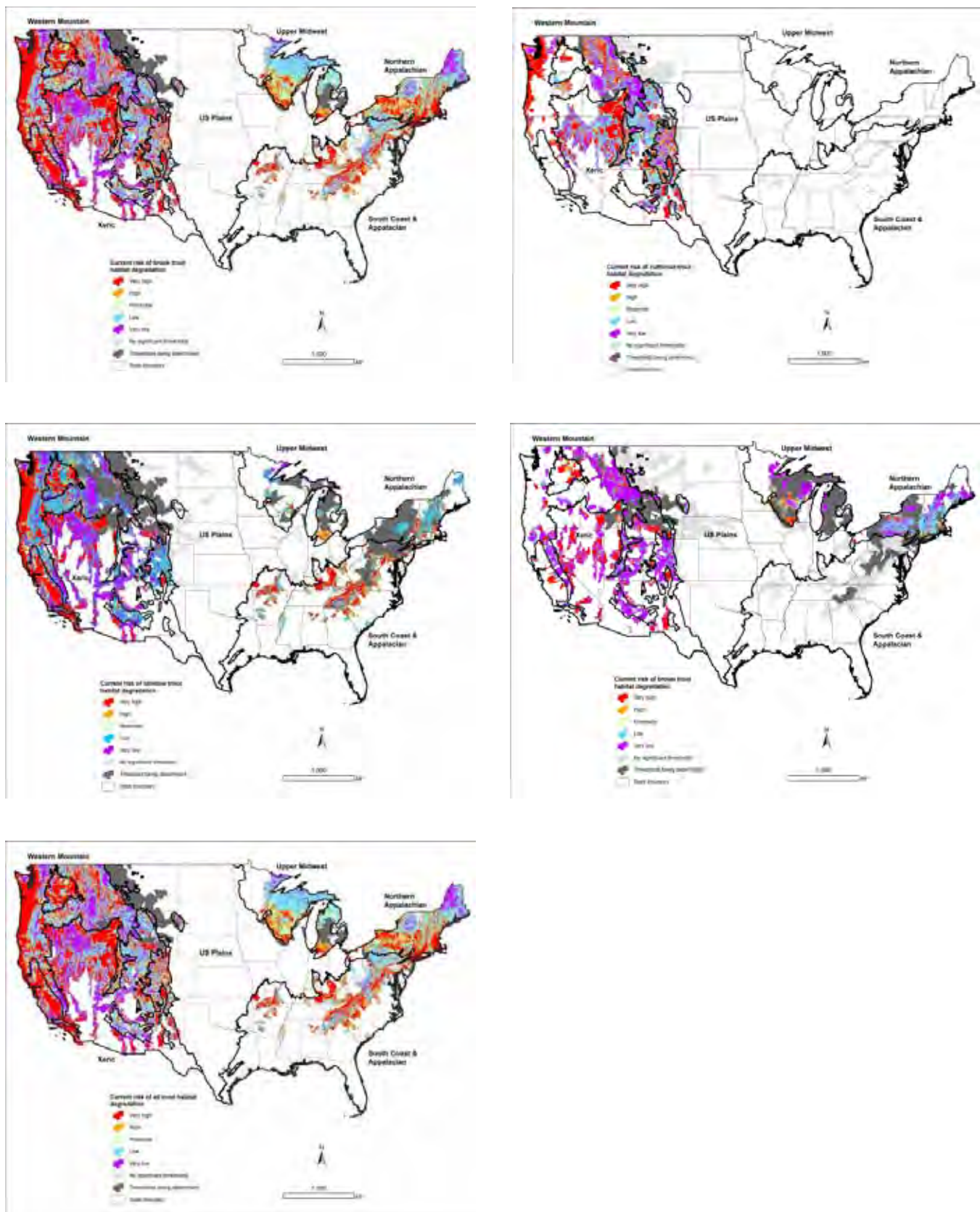


Figure 1. Current risk of habitat degradation for all watersheds analyzed by species and for all species combined.

Table 1. The most limiting disturbances to Cutthroat, Brown, Brook and Rainbow trout across their range and within the four spatial extents. First number is the percent (%) of stream kilometers with a pervasive (most common) landscape disturbance leading to a habitat condition score of (low, moderate, high or very high), and second number (in bold) represents the percent (%) of stream kilometers with that severe landscape disturbance leading to a habitat condition score of (high or very high). Grey areas represent landscape disturbance factors that were not tested within that spatial extent.

Brook Trout				
Disturbance Variables	Local catchment	Network catchment	Local Buffer	Network Buffer
Low urban	54.0 (95.7)	17.7 (10.2)	27.1 (20.2)	57.7 (41.5)
Med urban			2.9 (5.6)	
High urban		1.0 (1.5)	2.3 (4.1)	
Pasture		9.1 (7.1)	67.7 (70.2)	9.4 (22.0)
Crop		10.1 (9.8)		
Impervious				32.9 (36.5)
Road length		2.2 (5.9)		
Road cross		5.0 (9.5)		
Population	46.0 (4.3)	15.6 (15.6)		
M mine		0.6 (1.6)		
Dom. WW		1.6 (4.4)		
Ind. WW		4.1 (0.03)		
Total WW		2.4 (0.2)		
Down dams		9.9 (5.6)		
Nitrogen		3.4 (2.9)		
Phosphorous		6.9 (9.2)		
Sediment		10.4 (16.5)		
Brown Trout				
Disturbance Variables	Local catchment	Network catchment	Local Buffer	Network Buffer
Low urban		16.2 (6.2)		32.7 (1.0)
Pasture		3.2 (3.1)	100.0 (100.0)	15.3 (33.4)
Crop		21.2 (36.7)		25.5 (64.5)
Impervious				26.5 (1.1)
Road length	100.0 (100.0)	16.8 (2.3)		
Road cross		4.3 (9.5)		
Population		8.7 (4.3)		
Dom. WW		5.0 (4.8)		
Ind. WW		2.1 (3.3)		
Total WW		2.8 (4.9)		
Up dams		3.2 (2.4)		
Down dams		3.1 (5.2)		
Sediment		13.6 (17.3)		
Cutthroat Trout				
Disturbance Variables	Local catchment	Network catchment	Local Buffer	Network Buffer
Low urban	16.6 (15.8)	15.7 (24.3)		
Medium urban	3.5 (8.5)	1.5 (2.8)		
High urban		0.8 (0.5)		
Pasture	17.7 (36.9)		100.0 (100.0)	
Impervious surface				100.0 (100.0)
Road length	62.3 (38.8)	33.3 (39.3)		
Road crossings		14.8 (6.9)		
Population		3.5 (0.2)		
Uranium mine		3.0 (2.6)		
All mines		1.0 (1.5)		
Domestic WW		6.1 (0.7)		
Upstream dams		1.4 (3.3)		
Downstream dams		0.5 (1.1)		
Nitrogen		3.2 (3.7)		
Sediment		15.2 (13.1)		
Rainbow Trout				
Disturbance Variables	Local catchment	Network catchment	Local Buffer	Network Buffer
Low urban	14.6 (5.4)	5.6 (6.0)		22.0 (18.8)
Pasture		21.0 (28.8)		11.3 (22.9)
Crop		2.0 (7.2)		4.9 (7.6)
Impervious surface				61.8 (50.8)
Road length	57.3 (33.2)	3.4 (2.3)		
Road cross	10.5 (54.9)	27.2 (24.2)		
Population	17.6 (6.5)	1.4 (0.1)		
Domestic WW		4.6 (1.5)		
Total WW		10.7 (2.2)		
Upstream dams		0.8 (1.2)		
Nitrogen		15.0 (17.0)		
Sediment		8.3 (15.5)		

(10.5%) (Table 1). The two road factors, crossing density (54.9%) and length density (33.2%) were the majority (88.1%) of kilometers of severe landscape disturbances leading to habitat condition scores of high and very high disturbance risk for local catchments. The network catchment spatial extents had eleven significant pervasive and severe anthropogenic landscape factors associated with Rainbow Trout. The pervasive factors with >10% stream kilometers included road crossing density (27.2%), percent pasture land-use (21.0%), anthropogenic nitrogen yield (15.0%) and total water withdrawal (10.7%). These factors made up 73.9% of the significant thresholds. The most severe network catchment disturbances included road crossing density (24.2%), percent

pasture land-use (28.8%), nitrogen input (17.0%) and sediment input (15.5%). This group of disturbances accounted for 85.5% of the high and very high habitat conditions scores. Local buffer yielded no significant pervasive and severe anthropogenic landscape factors. The network buffer had four significant pervasive and severe anthropogenic landscape factors, with three factors having >10% of river kilometers: percent impervious surface being (61.8% pervasive and 50.8 severe), percent low-intensity urban land-use (22.0% and 18.8%) and percent pasture land-use (11.3% and 22.9%). Overall, the most important factors contributing to habitat disturbance risk for Rainbow Trout were road crossing density, pasture land-use, above background nitrogen yield and sediment input,

total water withdrawal, impervious surfaces, and low intensity urban land-use.

Brook Trout. – Of the four regions used in this study, all but the UPL region had significant Brook Trout threshold responses to anthropogenic landscape factors (Figure 1). A total of 15.9% of the threshold responses were significant, and 53.5% of those thresholds occurred in at the network catchment spatial extent. Based on the results of the assessment, 61.1% of kilometers of habitat for Brook Trout is at very low (22.4%) or low (38.7%) risk of habitat degradation from the tested anthropogenic landscape factors (Figure 2). This species also had combined (23.1%) of kilometers of habitats in the most degraded categories of very high (13.7%) and high (9.4%).

The local catchment scale yielded two significant anthropogenic landscape factors: percent low-intensity urban land use (54.0% pervasive and 95.7% severe) and human population density (46.0% pervasive and 4.3% severe) (Table 1). The network catchment spatial extent had fifteen significant pervasive and severe anthropogenic landscape factors. The most pervasive disturbance factors that accounted for >10% of stream kilometers included percent low-intensity urban land-use (17.7%), human population density (15.6%), anthropogenic sediment yield (10.4%) and percent crop land-use (10.1%). These factors made up 53.8% of the significant thresholds. When looking at the most severe disturbance factors, 42.3% of habitats with a high and very high risk of degradation condition were from anthropogenic sediment yield (16.5%), human population density (15.6%) and percent low-intensity urban land-use (10.2%). The local buffer spatial extent had four significant anthropogenic landscape factors but only two factors accounting for >10% of kilometers of streams: percent pasture land-use (67.7% pervasive and 70.2% severe) and percent low-intensity urban land-use (27.1% pervasive and 20.2% severe). The network buffer had three significant landscape factors, with two pervasive factors: percent low-intensity urban land-use (57.7%) and percent impervious surface (36.5%). Along with two severe factors: percent low-intensity urban land-use (41.5%) and percent impervious surface (36.5%). Overall, the most important factors contributing to habitat disturbance risk for Brook Trout were low intensity urban land-use and human population density, above background sediment inputs, crop and pasture land-uses, and impervious surfaces.

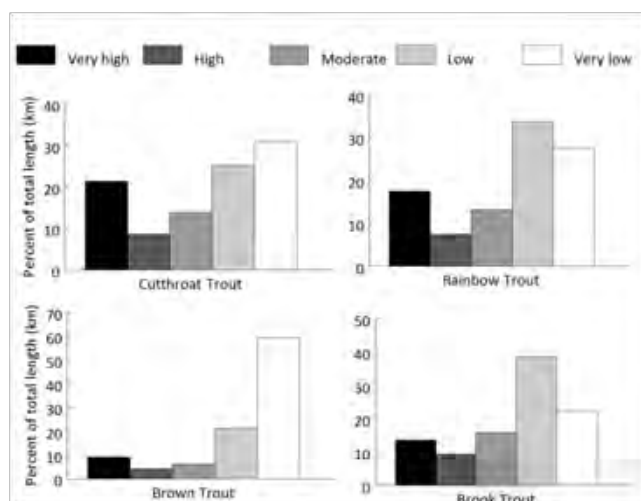


Figure 2. Relative condition of Cutthroat, Rainbow, Brown and Brook trout habitat in contiguous U.S. streams within their respective ranges using the percentage of total stream length in each condition class.

Brown Trout. – Of the six regions used in this study, only the NAP, UMW, WMT and XER regions had significant Brown Trout threshold responses to anthropogenic landscape factors (Figure 1). A total of 10.6% of the threshold responses were significant, and 69.6% of those thresholds occurred in the network catchment spatial extent. Based on the results of the assessment, 80.5% of Brown Trout stream habitat is at very low (59.4%) or low (21.1%) risk of habitat degradation from the tested anthropogenic landscape factors (Figure 2). That is the highest amount of the four trout species. This species also had the lowest combined kilometers of habitat (13.4%) at very high (9.2%) and high (4.2%) risk of degradation.

One significant anthropogenic landscape factor was documented at the local catchment scale, road length density (Table 1). The network catchment spatial extents had twelve significant pervasive and severe anthropogenic landscape factors. The pervasive factors that account for >10% of stream kilometers included: percent crop land use (21.2%), road length density (16.8%), percent low-intensity urban land use (16.2%) and anthropogenic sediment yield (13.6%). These factors made up 67.8% of the significant thresholds. At network catchment spatial scale, the most severe landscape factors were percent crop land use (36.7%) and anthropogenic sediment yield (17.3%) made up 54.0% of significant anthropogenic landscape

factors. The local buffer had a single significant pervasive and severe factor, percent pasture land use. The network buffer spatial scale had four significant pervasive landscape factors: percent low-intensity urban land use (32.7%), percent impervious surface (26.5%), percent crop land use (25.5%) and percent pasture land use (15.3%). When looking at the severe anthropogenic landscape factors in the network buffer spatial extent only percent crop land use (64.5%) and percent pasture land use (33.4%) accounted for >10% of stream kilometers. Overall, the most important factors contributing to habitat disturbance risk for Brown Trout were crop and pasture land-uses, road length density, low-intensity urban land-use, above background sediment inputs, and impervious surfaces.

Discussion

Study limitations. – Some important threats to trout and trout habitat that could not be incorporated into this assessment due to national dataset limitations as noted in Crawford et al. (2016). Based on missing factors and variables, trout habitat condition scores likely underestimate the true amount of disturbance in some systems and reaches and should be kept in mind as the reader interprets the information in this paper.

Another additional limitation was the use of fish community based sampling data from the National Fish Habitat Assessment which did not include the much more numerous species-focused sampling. Future assessments should use all available data to refine these initial habitat analyses.

The use of HUC8 watershed scale for the four trout species distribution likely overestimated the effects of some of the anthropogenic landscape factors as it captured waters within the watershed that are clearly not used by the trout species examined. Future analyses will use HUC12 scales which should eliminate the variability created by the inclusion of obvious non-trout water and will improve the sensitivity of the analysis.

Cutthroat Trout. – Pervasive and severe anthropogenic landscape factors associated with habitat degradation for Cutthroat Trout were driven by road length and crossings, urban and agricultural land use, and nutrient and sediment loadings. These findings follow those of a recent assessment by Williams et al. (2015), where road development, urban land use, and agricultural land use were common reasons for Cutthroat Trout declines throughout their native

range. Studies examining effects of anthropogenic landscape variables on Cutthroat Trout over large spatial extents are limited in number, but those that have been conducted emphasize landscape characteristics and the spatial extent of those characteristics are important to understanding distribution, abundance, and conservation status of Cutthroat Trout (Gresswell et al. 2006; Williams et al. 2015). A high percentage of road crossings are culverts, often higher than 60%, and many culverts, often higher than 60%, are impassible by fish, fragmenting river networks and isolating populations. Isolation has been documented to reduce Cutthroat Trout populations by preventing immigration, emigration and expression of migratory or mobile life history (Fausch et al. 2006), reducing genetic diversity (Pritchard et al. 2007), and reducing habitat space, complexity, and variability (Dunham et al. 1997). The effects of fragmentation on isolated Cutthroat Trout populations can be further compounded by environmental variation, natural or anthropogenic, in the constrained habitat (Hildebrand and Kershner 2000).

Additionally, Cutthroat Trout had the highest percentage of degraded habitat of the four species tested. In degraded habitat, Cutthroat Trout species are more likely to be displaced by or hybridize with trout species nonnative to the area, such as Brook, Brown, and Rainbow trout (Dunham et al. 2002; Shepard 2004; Muhfield et al. 2009; Williams et al. 2015). This suggests that Cutthroat Trout are more sensitive to anthropogenic landscape factors than the other three trout species tested in this study. Use of barriers to prevent invasion of nonnative salmonids into Cutthroat Trout inhabited streams is one management practice that has been implemented but results in further fragmentation and degradation of Cutthroat Trout habitat (Shepard et al. 2005).

Rainbow Trout. – Pervasive and severe anthropogenic landscape factors associated with habitat degradation for Rainbow Trout were driven by urban land use, road density with similar effects to those noted for Cutthroat Trout, pasture, nitrogen and sediment inputs, and water withdrawals. These findings are similar to those discussed in Williams et al. (2015). There are few landscape scale studies on Rainbow Trout to compare these results. The WMT (native range) and SCA (introduced range) both show a large amount of very high risk for degradation of habitat with much of the rest of the country under examination at this time. While Rainbow Trout had the

second highest percentage of very high risk waters, this species appears to be intermediate in overall degraded habitat risk with over 60% of the habitat with very low or low habitat degradation risk. This is likely a result of this species being broadly tolerant of anthropogenic alteration.

Brook Trout. – Pervasive and severe anthropogenic landscape factors associated with habitat degradation for Brook Trout were driven by urban land use, impervious surface, sediment yield and population density key factors. These findings are similar to those in more regionally or state based analyses which were generally in the native Brook Trout range (Pennsylvania: DeWeber and Wagner 2015; Eastern U.S.: Hudy et al. 2008; New York: McKenna and Johnson 2011; Michigan: Siitari et al. 2011 and Steen et al. 2010; Maryland: Strank et al. 2008). Overall, the risk of Brook Trout habitat degradation was often higher in the WMT and XER (nonnative parts of their range), although there were clear large patches of low risk habitat in these ecoregions, than in the northern parts of this species native range (NA and UMW). This is likely an artifact of the use of the HUC8 scale that will be investigated further by the authors. Brook Trout are always considered to be a very sensitive species to environmental change as noted in Hudy et al. (2008) and Steen et al. (2010) along with many other studies but our results show them intermediate in frequency of degraded habitat when compared to the other trout species which may be an artifact of the spatial scale used in this study.

Brown Trout. – Pervasive and severe anthropogenic landscape factors causing Brown Trout habitat to be degraded were related to high numbers of stream-road crossings, urban land use, impervious surface, agricultural land, and nutrient and sediment loadings. However, a majority of Brown Trout habitat in the NAP, UMW, WMT, and XER regions was at low or very low risk of habitat degradation, the highest amount of good habitat across species tested, and likely from a higher tolerance to anthropogenic landscape factors. They have a greater tolerance to ecological variability than other trout species as expressed by: the occupancy of diverse habitats, a broad array of life history strategies (e.g., migratory, sedentary, territorial), and the ability to withstand greater water temperature and quality variability (McIntosh et al. 2011). As a result of these broader tolerances, they often replace or displace native trout

species including Brook, Cutthroat, and Rainbow trout, particularly in altered trout habitat (Williams et al. 2015).

Overall Study Conclusions. – Of the four species of salmonids examined in this analysis, Brown Trout showed the least number of landscape impairments and as expected, are likely the most tolerant to anthropogenic landscape factors. Similarly, Cutthroat Trout appeared to be the least tolerant of this group of salmonids. The network scale appeared to much more sensitive and found many more signals from anthropogenic factors than the local catchment scale. Key anthropogenic factors were found at each scale: (1) local catchment - low-intensity urban land-use, road length density, and human population density; (2) network catchment - low-intensity urban land-use, and road length and crossing density, sediment input, and crop and pasture land-uses; (3) local buffer - pasture land-use, and low-intensity urban land-use; and (4) network buffer - low-intensity urban land-use, impervious surface, and pasture land-use. The effects of urbanization, increasing road densities and crossings and pasture land use are clearly shown in the abundance of signals provided by the four trout species with some indications of issues from sediment inputs, impervious surface, and row cropland.

Our initial study indicates that it is possible to access trout habitat data across the United States, even with the limitations in the national dataset used, and this is the first study to attempt to draw a national picture for this economically important set of recreationally fished species. Overall, national results were generally consistent with more localized or regional studies done previously. The addition of a broader range of environmental conditions and variables with a much larger sample size using both native and introduced ranges of these species, likely increases the strength of the models when compared to studies done within more limited geographic areas. We expect that rapid improvements will be made in our models once the noted data limitations are addressed. The developed relationships in this study can be used to examine habitat condition, as defined by risk of degradation, for individual and mixed groups of trout, determine appropriate conservation measures to improve a range of trout species, and can be used as predictive metrics to examine systems with little fisheries survey information, particularly when abiotic variability is statistically controlled.

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Stewardship and Restoration on the White River, CO: Legacy and Novel Restoration Techniques

Josh Epstein

Inter-Fluve, Inc., Hood River, Oregon

Abstract—River and floodplain conditions that influence native fish are an expression of geology, climate, biology, and the many consequences of the Anthropocene. A legacy of stewardship and preservation by a large ranch on the upper White River, Colorado, has resulted in multiple phases of human interventions in the channel and floodplain. A continuation of legacy practices in the region, boulder drop structures were installed decades ago to create pool habitats. A constructed highly sinuous secondary channel created new hydrologic connections with a portion of the floodplain. The constructed project altered ecological and geomorphic processes resulting in homogenous aquatic habitat and a secondary channel that had an unsustainable level of sinuosity. The structures served as impediments to channel and sediment processes and associated pools filled with sediment over time resulting in habitats dominated by a plane-bed-channel type. Seeing that their river channel was out of balance, the ranch owners sought to re-naturalize a 3-mi reach of the White River. As native fish species require a diversity of aquatic habitats during different life stages, the White River restoration project utilized a natural process approach that involved the removal of structures and the restoration of a more geomorphically appropriate channel. The project included removal of rock structures, decommissioning of the sinuous secondary channel, wetland creation, and riparian floodplain restoration. The project has restored aquatic and floodplain diversity influencing both aquatic and terrestrial species, including native salmonids. While the previously constructed project reflected an intention of making a positive influence on the aquatic ecosystem using legacy technologies and practices, the rock structures did not fit the natural process context and geomorphic setting at the site. Novel techniques that emphasize natural processes are informed by historical and future trends for a design solution.





Evaluating Trout Stream Restoration Benefits: A Case Study at Pine Creek, Wisconsin

Kent Johnson

Aquatic Biologist, Kiap-TU-Wish Chapter, Trout Unlimited, Hudson, WI

Abstract—In 2007–2011, the Wisconsin Department of Natural Resources and the Kiap-TU-Wish Chapter of Trout Unlimited conducted an extensive stream restoration project at Pine Creek, a native Brook Trout *Salvelinus fontinalis* stream in the Driftless Area of Wisconsin. Primary project objectives were to remedy severe stream bank erosion and increase Brook Trout abundance by 40–50%. The project restored 2.11 stream miles at a cost of \$270,000. In 2009, the Pine Creek Restoration Project was recognized by the National Fish Habitat Action Plan as one of 10 national “Waters to Watch”. Key elements of a monitoring program to evaluate project success included physical and biological attributes measured pre- and post-restoration. Physical attributes included stream temperature and habitat (stream width, water depth, water velocity, canopy cover, stream bank height and cover, and stream bed substrate). Biological attributes included macrophytes, macroinvertebrates, and trout. Beneficial project outcomes included a decrease in stream temperature, a reduction in stream width, greatly reduced stream bank heights and erosion potential, and increases in water depth, stream bank cover, presence of coarse stream bed substrate, and macrophyte presence. Unanticipated project outcomes included decreases in canopy cover and water velocity, no significant improvement in macroinvertebrate metrics, and a significant increase in Brown Trout *Salmo trutta* abundance and decrease in Brook Trout abundance. Within 8 years post-restoration, numbers of Brook Trout per mile decreased by 70% (3,800 to 1,200), while numbers of Brown Trout per mile increased by 3,150% (175 to 5,600). A continuation of this trend may lead to the loss of the Brook Trout fishery. With Brook Trout being the only native trout species in the Driftless Area, this project highlights the need for appropriate restoration techniques that can protect and enhance Brook Trout in streams that could be subject to Brown Trout cohabitation.

Introduction

Pine Creek is a third-order, coldwater stream located in Maiden Rock Township, Pierce County, Wisconsin, at the northwestern end of the Upper Midwest’s Driftless Area (Figure 1). Pine Creek emanates from a series of large springs and flows westerly into the Mississippi River at Lake Pepin. Consisting primarily of heavily forested coulees and upland agricultural areas, the Pine Creek watershed is part of the karst landscape of the Driftless Area ecoregion, which is characterized by thin loess soils underlain by fractured limestone. The Wisconsin Department of Natural Resources (WDNR) lists Pine Creek as a Class I trout stream that has historically sustained a naturally-reproducing population of Brook Trout *Salvelinus fontinalis*. Approximately 1.8 miles of Pine Creek and 1.1 miles of classified tributaries are protected in the Pine Creek Fishery Area. The lower 2 miles of Pine Creek are separated from the headwaters by approximately 0.5 mile of subterranean flow (WDNR 2017).

As is characteristic of many streams in the Driftless Area, Pine Creek has good water quality but has suffered from severe stream bank erosion. In the early 20th century, poor agricultural practices and runoff from the watershed mobilized the thin loess soils at the tops of the surrounding bluffs and deposited them in the valley floor. Before a stream restoration project began in 2007, Pine Creek was still moving these deposits, resulting in steeply eroded and raw stream banks with massive deposition of fine sediment in the streambed. Overgrazing on adjoining pasture lands compounded the erosion and sedimentation problem, which severely limited habitat and Brook Trout reproduction in Pine Creek.

In 2002 and 2003, the West Wisconsin Land Trust (WWLT) purchased two agricultural properties (220 acres) that encompass the majority of the Pine Creek Fishery Area, thus conserving these properties forever. With much of the stream corridor in WWLT ownership and open to the public, WWLT, three Trout Unlimited



Figure 1. Location of Pine Creek in the Upper Midwest's Driftless Area

(TU) chapters, and WDNR began planning a stream restoration project in 2006. The goal of the Pine Creek Restoration Project was to enhance and conserve the native Brook Trout population in Pine Creek by stabilizing severely eroding banks, providing in-stream cover, and improving aquatic habitat in the stream. Measurable project objectives included:

1. Improve the stream temperature regime and armor the stream for climate change.
2. Reduce stream bank erosion to 10% of pre-existing conditions.
3. Increase coarse stream bottom substrate by 50%.
4. Increase aquatic macrophyte growth by 25%.
5. Increase numbers of Brook Trout by 40-50%.
6. Increase numbers of Brook Trout 10 inches and larger by 50-100%.

During the 2007-2011 period, 2.1 miles of Pine Creek and two major spring tributaries were restored by WDNR, in partnership with TU (Kiap-TU-Wish, Clear Waters, and Twin Cities Chapters), WWLT, Fairmount Santrol, the National Fish Habitat Action Plan (NFHAP), the U.S. Fish and Wildlife Service, the Trout and Salmon Foundation, and Patagonia (Sours 2011). The total cost of the Pine Creek Restoration Project was \$270,000 (\$24 per lineal foot of stream). In 2009, the success of the project was recognized by the NFHAP, which listed Pine Creek as one of 10 national "Waters to Watch".

Stream restoration is an integral part of trout stream management in Wisconsin, with the restoration work generally targeting Class I or

Class II trout streams. Although stream restoration may take different forms, it generally involves the re-establishment of aquatic functions and related biological, chemical, and physical characteristics of streams that would have occurred prior to disturbance. Trout anglers fishing inland waters in Wisconsin are required to purchase a trout stamp, from which the proceeds are directed toward stream habitat restoration work. Hunt (1988) and Avery (2004) have documented a half century (1953-2000) of evaluations of trout stream habitat restoration projects in Wisconsin, and have shown how restoration has been successful at improving trout populations in terms of trout number and size (Mitro et al. 2011).

The Pine Creek Restoration Project was accomplished using techniques developed by WDNR fisheries managers across the Driftless Area (White and Brynildson 1967; Hunt 1993). Steep eroding banks were sloped back (typically at a 3:1 slope) to open the stream channel to the flood plain, thereby dissipating flood energy. As a result, stream bank erosion and sedimentation are greatly diminished, water can infiltrate in the riparian area, and water pollutants can be removed and processed. Where suitable, "LUNKER" structures were added to provide trout cover from predators and refuge during floodwaters (Vetrano 1988). These structures were covered with rock and soil and then reseeded to stabilize the stream banks. Boulder clusters and root wads were installed to enhance midstream cover. In addition, plunge pools were excavated to create deep water and overwintering habitat. The installation of bank cover narrows the

stream, which results in bottom scouring that exposes gravel substrate favorable for aquatic insects and successful trout reproduction. Bank stabilization results in a decrease in suspended sediment during runoff events, thus improving water quality in the stream. An improvement in the temperature regime of the stream may also occur, due to a narrower, deeper channel, increased current velocity, and bank shading.

With degraded coldwater streams present throughout the Driftless Area, and with global climate change posing an increasing threat to these sensitive systems (Mitro et al. 2011), stream restoration is a critical tool for enhancing and protecting aquatic ecology, and upland restoration is an effective means of improving water quality and sequestering carbon. With limited resources available, it is imperative that restoration practices produce the best long-term outcomes with the most efficient use of funding, for ecological and public benefits.

Methods

Because of the cost and visibility of the Pine Creek Restoration Project, it was very important to document

the achievement of the project objectives, as presented above. This was accomplished by measuring pre- and post-restoration temperature and habitat conditions, trout densities and size distribution, and macrophyte and macroinvertebrate community health. Evaluation of the project objectives was conducted jointly by local WDNR fisheries staff and Trout Unlimited (Kiap-TU-Wish Chapter) volunteers, via collection of pre- and post-restoration temperature, habitat, and biotic data. The methods used for pre- and post-restoration monitoring of Pine Creek air and stream temperatures, habitat conditions, macrophyte presence, and macroinvertebrate communities are described in Hastings, et al. (2011) and Johnson (2017).

Hunt (1971) has emphasized the critical need to document quantitative changes in trout populations and their environment as a result of stream restoration. At Pine Creek, WDNR fisheries staff have been conducting trout surveys at two sites within the restoration project area (Figure 2), using WDNR monitoring protocols for coldwater wadeable streams (Lyons et al. 1996; WDNR 2007). To conduct the survey work, WDNR staff use a stream barge



Figure 2. WDNR's Pine Creek trout survey sites

electrofishing unit with 3 electrodes. The generator runs DC at 100-200V and 4A. The survey crew consists of the three electrode handlers/netters and a boat operator/puller. The survey station length is 35 times the mean stream width, which was calculated pre-restoration and continues to be used post-restoration. Surveys are conducted on a catch per unit effort basis, using one pass in an upstream direction. Effort time is recorded but trout numbers are generally compared by distance (number per mile). Air and water temperatures and weather conditions are also recorded on the day of the survey. Any extenuating circumstances (flooding, turbidity, excessive plant growth, etc.) which may have an effect on the catch rate are also noted. All trout surveys are conducted between June 15 and September 15, to allow capture of young-of-year fish.

Trout surveys in the upper part of the restoration area (Station 2A) were conducted in 2000, then annually during the 2005-2017 period. Pre-restoration survey years at Station 2A include 2000 and 2005-2008. Post-restoration survey years at Station 2A

include 2009-2017. The stream length surveyed at Station 2A was 219 yards. Trout surveys in the lower part of the restoration area (Station 2B) were conducted during the 2005-2010 period. Pre-restoration survey years at Station 2B include 2005-2007, while post-restoration survey years at Station 2B include 2008-2010. The stream length surveyed at Station 2B was 188 yards. Survey data included Brook Trout and Brown Trout numbers and lengths. Based on the stream distance surveyed at each station, numbers of Brook Trout and Brown Trout per mile were estimated for young-of-year fish, adult fish in multiple size categories (typically 1-in to 2-in length increments), and all size categories combined (total trout per mile). The WDNR trout survey data were used to determine whether Project Objectives 5 and 6 were met.

Results

The results of pre- and post-restoration monitoring of Pine Creek stream temperatures, habitat conditions, macrophyte presence, and macroinvertebrate

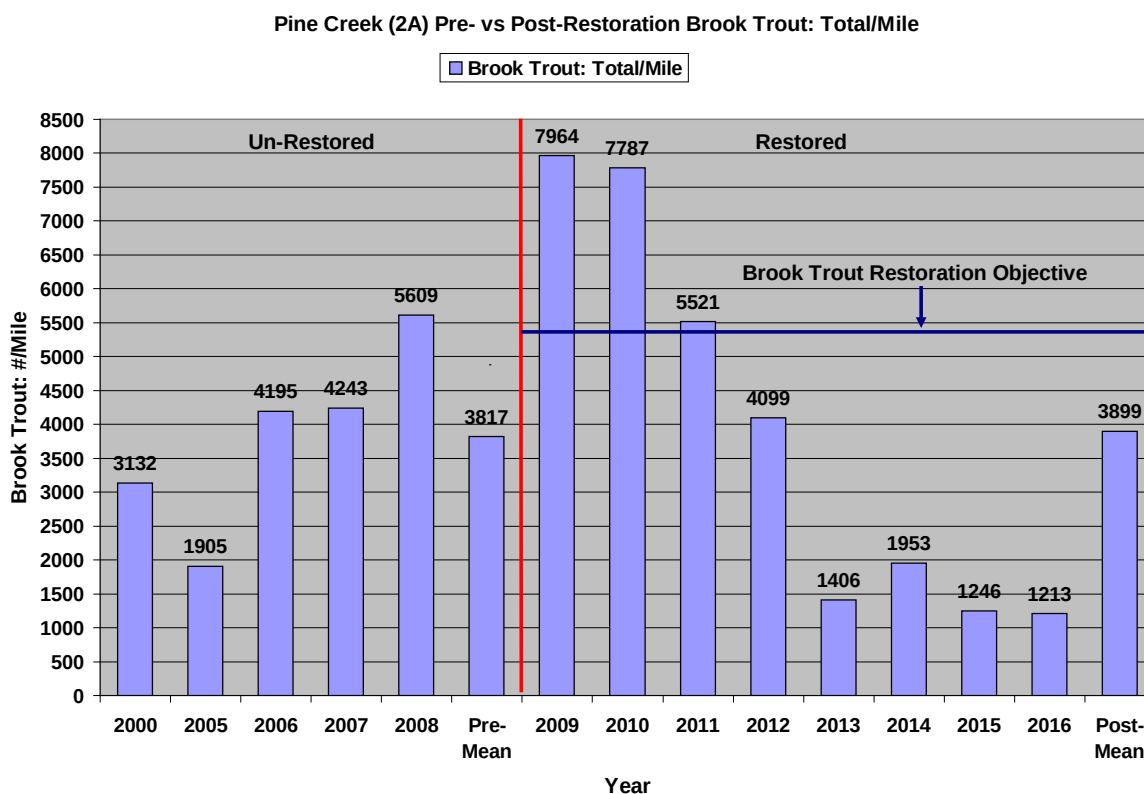


Figure 3. Pre- and post-restoration abundance of Brook Trout in Pine Creek at Station 2A

communities are presented in Johnson (2017). Improvement of the native Brook Trout fishery was a primary focus of the Pine Creek Restoration Project, as noted in Project Objectives 5 and 6. Since WDNR Station 2A in the upper part of the restoration area (Figure 2) has the best record of annual trout survey data, this station can be used to compare the pre- and post-restoration trout populations. A caveat of the survey data at Station 2A is the assumption that this station is representative of trout abundance and size in the remainder of the restoration reach.

A comparison of the pre- and post-restoration abundance of Brook Trout in Pine Creek (expressed as total trout/mi) is shown in Figure 3. The pre-restoration abundance of Brook Trout in Pine Creek was already robust, ranging from 1,905-5,609 trout/mi and averaging 3,817 trout/mi during the 5-year pre-restoration survey period. The Brook Trout population immediately benefited from the restoration work, with post-restoration abundance increasing dramatically to 7,787-7,964 trout/mi in 2009-2010. In subsequent years, however, Brook Trout abundance in Pine Creek

has experienced a steep decline, reaching a minimum of 1,213 trout/mi in 2016. As of 2016, Brook Trout abundance has decreased by 68%, compared to mean pre-restoration abundance. Project Objective 5 targeted a 40-50% increase in Brook Trout numbers.

A comparison of the pre- and post-restoration abundance of 10-in+ Brook Trout in Pine Creek (Figure 4) shows a similar trend. Annual pre-restoration abundance of these larger Brook Trout varied widely, ranging from 0 to 72 trout/mi and averaging 31 trout/mi during the 5-year pre-restoration survey period. After a rapid post-restoration increase that peaked at 104 trout/mi in 2011, the abundance of 10-in+ Brook Trout has declined dramatically, reaching a minimum of 0 trout/mi in 2015. As of 2016, the abundance of 10-in+ Brook Trout in Pine Creek has decreased by 74%, compared to mean pre-restoration abundance. Project Objective 6 targeted a 50-100% increase in 10-in+ Brook Trout numbers.

While the post-restoration abundance of Brook Trout in Pine Creek has been rapidly decreasing, the post-restoration abundance of Brown Trout has

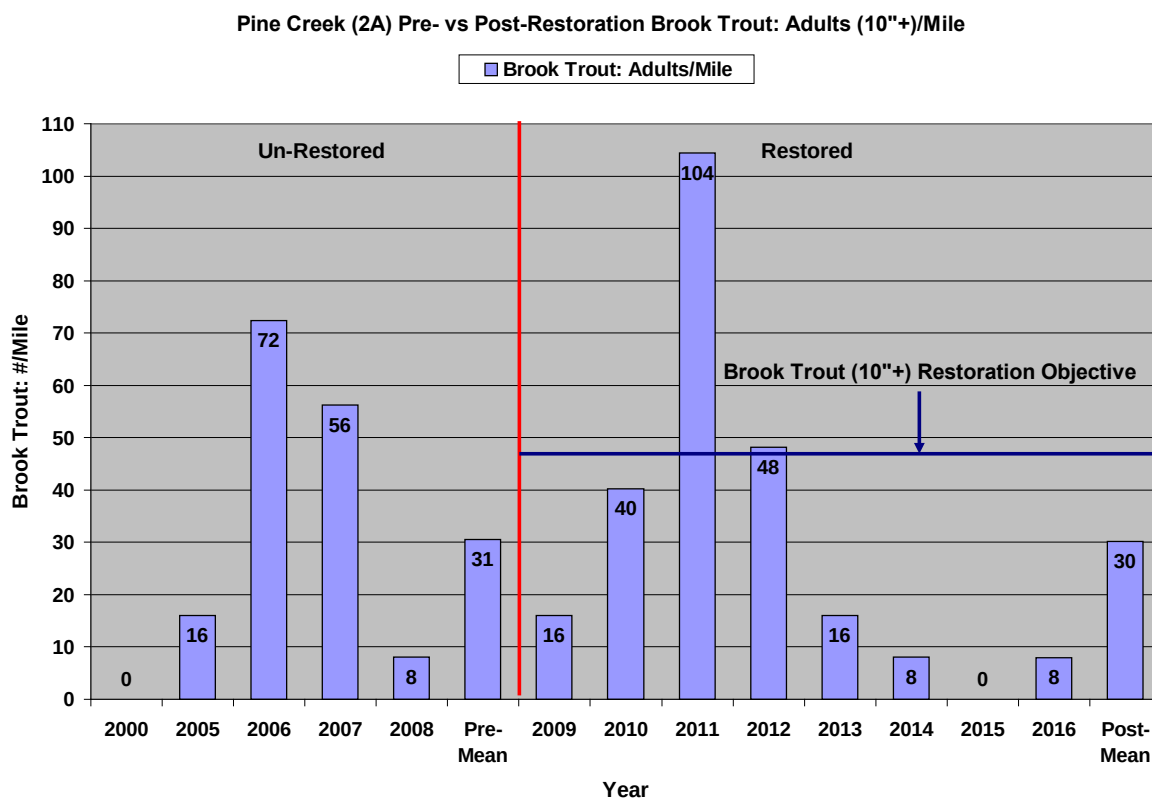


Figure 4. Pre- and post-restoration abundance of 10-in+ Brook Trout in Pine Creek at Station 2A.

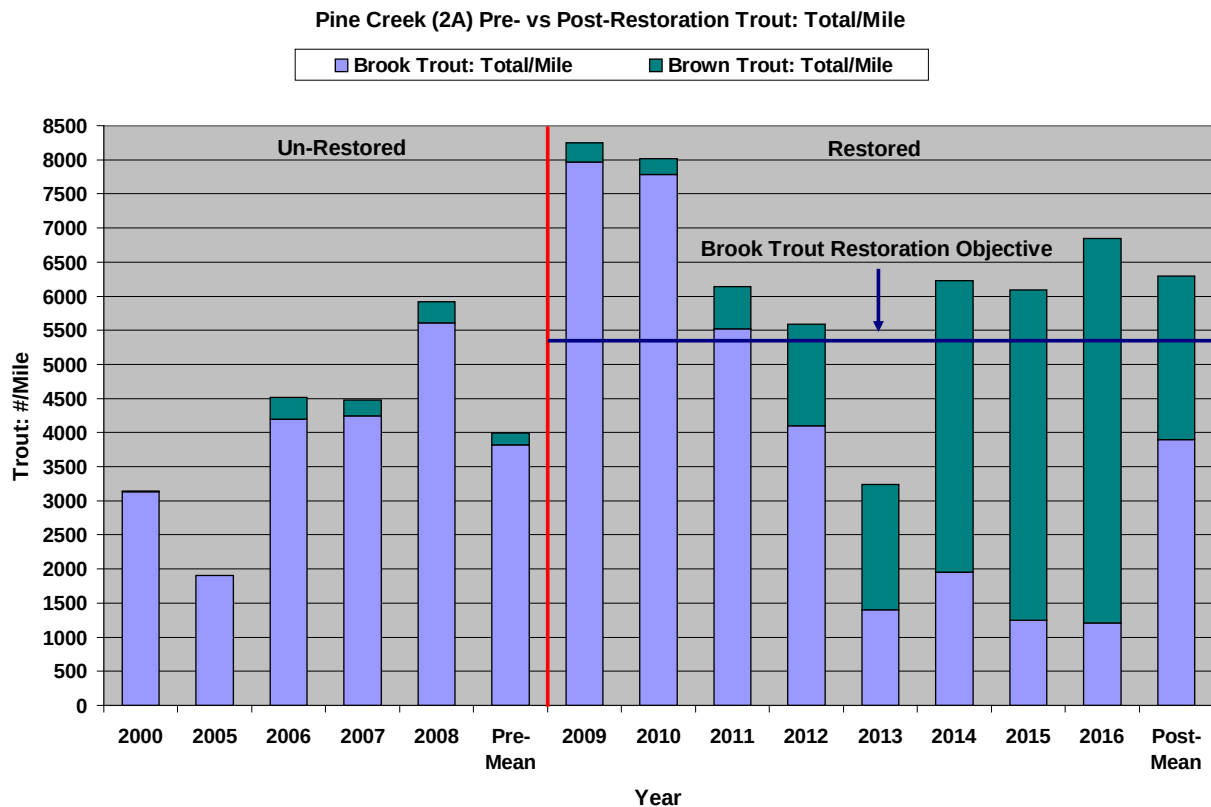


Figure 5. Pre- and post-restoration abundance of Brook Trout and Brown Trout in Pine Creek at Station 2A.

increased markedly (Figure 5). Small numbers of Brown Trout were present in annual pre-restoration surveys at Station 2A, but the Brown Trout proportion of total trout abundance never exceeded 7%, with the stream dominated by Brook Trout. The post-restoration abundance of Brown Trout in 2009-2010 remained similar to the pre-restoration abundance. However, a steep increase in Brown Trout abundance began in 2011, with the greatest increase occurring between 2013 and 2014. In 2016, Brown Trout abundance in Pine Creek reached 5,633 trout/mi, representing a 3,137% increase, compared to mean pre-restoration abundance. On average, pre-restoration trout abundance in Pine Creek was 3,991 trout/mi, with Brook Trout and Brown Trout present in a 96%:4% proportion. In comparison, post-restoration trout abundance has averaged 6,299 trout/mi, with Brook Trout and Brown Trout present in a 62%:38% proportion. However, with rapidly-increasing numbers of Brown Trout in Pine Creek since 2011, the proportion of Brook Trout has decreased to 18% in

2016. Comparing mean pre-restoration to mean post-restoration trout abundance (Figure 5), the Pine Creek Restoration Project has resulted in a 58% increase in the Pine Creek trout population. However, this increase is due to the substantial expansion of Brown Trout presence in Pine Creek.

Discussion

Pre- and post-restoration monitoring of stream temperature, habitat, and biota was an integral part of the Pine Creek Restoration Project, providing a wealth of information on project outcomes, including benefits, unintended consequences, and opportunities for improvement. Monitoring also enabled a determination of whether the six key project objectives were met (Johnson 2017). A discussion of project benefits for improving the Pine Creek temperature regime, greatly reducing erosion potential, and re-connecting the stream to the floodplain can also be found in Johnson (2017). In summary, the restoration project resulted in a measurable improvement in the stream temperature

regime (Project Objective 1), a 60% decrease in stream bank height (Project Objective 2), a 140% increase in coarse stream bottom substrate (Project Objective 3), and a 133% increase in aquatic macrophyte presence (Project Objective 4). Other habitat changes included a 40% decrease in stream channel width, a 75% increase in water depth, a 15% decrease in stream velocity, and a 20% decrease in canopy cover.

The main impetus for the Pine Creek Restoration Project and the primary project goal was to enhance and conserve the native Brook Trout population in Pine Creek. However, within 5 years post-restoration, Pine Creek had become dominated by Brown Trout (Figure 5), a significant unanticipated consequence of the restoration project. By 2013, Brook Trout abundance in Pine Creek was lower than that during any of the 5 pre-restoration years surveyed, and abundance has continued to decline through 2016. This outcome represents a dramatic reversal of Brook Trout presence in Pine Creek, falling far short of Project Objective 5, a 40-50% increase in Brook Trout numbers. Brown Trout were already present in Pine Creek before the restoration project began (Engel, personal communication, 2017). However, Brown Trout abundance was very low, ranging from 233 to 321 trout/mi during the pre-restoration period of 2006-2008 (Figure 5). In comparison, Brook Trout abundance ranged from 4,195-5,609 trout/mi during the same period, with Brook Trout comprising 94% of the Pine Creek trout population.

Although the pre-restoration abundance of Brown Trout in Pine Creek was very low (6%), WDNR was concerned about their presence in a Class I, naturally-reproducing Brook Trout stream. As a result, WDNR trout survey crews attempted to purge Pine Creek of Brown Trout via electrofishing and removal in 2007 and 2008. However, trout surveys in 2009 and 2010 (Figure 5) showed that this effort was unsuccessful, and Brown Trout removal was no longer a viable management option as post-restoration Brown Trout abundance increased rapidly (Engel, personal communication, 2017).

Engel (personal communication, 2017) notes that the post-restoration success of Brown Trout in Pine Creek may be due in part to their ability to out-compete Brook Trout for occupation of the best available habitat, which the restoration project created via installation of LUNKER structures, boulder clusters, and root wads. Fausch and White (1981) note

that Brown Trout exclude Brook Trout from preferred resting positions, a critical and scarce resource. The combined effects of such interspecific competition, an increased susceptibility of Brook Trout to angling, differential response to environmental factors, and Brown Trout predation on juvenile Brook Trout may account for declines of Brook Trout populations while Brown Trout populations expand in many streams where the two species coexist. Hitt et al. (2017) note that the distribution of native Brook Trout in eastern North America is often limited by Brown Trout, in part via interference competition for access to thermal refugia and forage habitats. Waters (1983) observed that the trout population in Valley Creek (MN) changed over 15 years from virtually 100% Brook Trout to predominantly Brown Trout, due to innate factors in the behavior of Brook Trout and Brown Trout and interactions with habitat perturbations.

The post-restoration increase in overhead cover and shade provided by LUNKER structures and root wads may also favor the presence of Brown Trout in Pine Creek. Cover is recognized as one of the basic and essential components of trout streams, as noted by Boussu (1954), Lewis (1969), and Raleigh (1982). In a study to determine the amount of shade utilized by Brook Trout, Rainbow Trout *Oncorhynchus mykiss*, and Brown Trout, Butler and Hawthorne (1968) reported that Rainbow Trout showed the lowest preference for shade produced by artificial surface cover. Brown Trout showed the highest use of shade, while Brook Trout were intermediate between Brown Trout and Rainbow Trout.

Engel (personal communication, 2017) believes that habitat restoration in Brook Trout streams will result in improved Brook Trout populations and size structure. However, if Brown Trout have access to these streams, Brown Trout will prevail but not totally eliminate Brook Trout. The dramatic post-restoration change in Pine Creek trout dynamics suggests that trout stream restoration in the Driftless Area should not be a “one size fits all” exercise. An exceptionally cold temperature regime in Pine Creek did not provide a competitive advantage for Brook Trout, and Brown Trout removal was unsuccessful, even when abundance was low.

Resource managers hoping to protect and enhance native Brook Trout streams, especially those vulnerable to Brown Trout cohabitation, should consider an adaptive management approach

that creates habitat favorable for Brook Trout. This consideration will become even more critical as climate change creates stream temperature regimes that are more suitable for Brown Trout, at the expense of Brook Trout (Mitro et al. 2011; Cunningham, et al. 2014).

Acknowledgments

The WDNR trout restoration team (John Sours and Nate Anderson) deserves credit for transforming Pine Creek. Trout Unlimited volunteers from the Kiap-TU-Wish, Clear Waters, and Twin Cities Chapters and Fairmount Santrol staff provided outstanding support for the restoration work. WDNR fisheries staff (Marty Engel and Barb Scott) conducted trout surveys and provided valuable data and interpretation for understanding trout dynamics. The Kiap-TU-Wish monitoring team (Andy Lamberson, John Kaplan, Dave Cumming, and Scott Wagner) provided excellent support for temperature, habitat, and macrophyte monitoring. Kiap-TU-Wish project manager Greg Dietl deserves special recognition for his project role, serving as the liaison with the WDNR trout restoration team, coordinating volunteer support, and writing grants for project funding. The financial support provided by the National Fish Habitat Action Plan (NFHAP), the U.S. Fish and Wildlife Service, the Trout and Salmon Foundation, and Patagonia helped to make this project possible. Finally, the West Wisconsin Land Trust (WWLT) deserves recognition for purchasing two properties that encompass the majority of the Pine Creek Fishery Area, thus conserving these properties forever and making the restoration project possible.

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White, R.J. and O.M. Brynildson. 1967. Guidelines for management of trout stream habitat in Wisconsin. Technical Bulletin Number 39. Wisconsin Department of Natural Resources, Madison. Response of Wild Brook Trout and Rainbow Trout Populations to Physical Habitat Enhancement Projects Designed to Favor Brook Trout





Response of Wild Brook Trout and Rainbow Trout Populations to Physical Habitat Enhancement Projects Designed to Favor Brook Trout

K. M. Kuhn

Area Fisheries Manager, Pennsylvania Fish and Boat Commission, 844 Big Spring Road,
Newville, PA 17241; kkuhn@pa.gov

Abstract—Big Spring Creek, a Pennsylvania alkaline spring stream, was historically renowned for its Brook Trout *Salvelinus fontinalis* fishery. Currently, sympatric populations of Brook Trout, Rainbow Trout *Oncorhynchus mykiss*, and Brown Trout *Salmo trutta* reside in the stream. Considering ecological, social and economic factors associated with native species management, the management strategy applied to this stream was modified to treat the stream as a sympatric salmonid fishery, but to manage preferentially for Brook Trout. Habitat enhancement projects were completed in stream reaches dominated by Rainbow Trout; however, projects were designed to replicate habitat that favors Brook Trout. Since completion, Brook Trout have increased while Rainbow Trout have declined following an initial proliferation of that species posttreatment. Preliminary results indicate the response to the projects is trending toward a Brook Trout dominant salmonid community. Habitat manipulations to bolster a Brook Trout population in the presence of nonnative salmonids is a novel approach in a Pennsylvania spring stream, and these results provide insight regarding efficacy of this technique as a mechanism towards native species conservation. However, this approach will likely need to be coupled with a nonnative removal mechanism such as selective harvest to increase the likelihood that Brook Trout remain the dominant salmonid over the long term.

Introduction

In the United States, more than 500 exotic fish taxa have been introduced through anthropogenic activities to areas or ecosystems apart from their historic geographic range. Many of these nonnative fishes have become established resulting in United States inland-water fish assemblages drastically altered from their pre-European settlement condition (Nico and Fuller 1999). Widespread habitat alterations and the repeated reintroduction of nonnative fishes, the majority of which were introduced to enhance recreational fisheries, have facilitated the proliferation of these fishes throughout the United States (Gido and Brown 1999).

In many cases, nonnative species introductions, whether intentional or unintentional, have resulted in decline, extirpation, or extinction of native fishes through direct competition for limited resources and predation. Nonnative fishes that become naturalized are frequently attributed to declines of native fish populations and substantially limit success of native fish species restoration efforts (Sheldon 1988; Miller et al. 1989; Richter et al. 1997; Wilcove et al. 1998). Naturalized nonnative species, however, can provide

valuable sport fisheries and are commonly stocked in waters outside their native range, providing local and regional recreational and economic benefit. Due to habitat degradation and declining native fish assemblages, fisheries management agencies must decide among compromises regarding game, nongame, native, and nonnative species management (Beamesderfer 2000). Fisheries management agencies must also decide whether nonnative game fish management is compatible with native fish restoration efforts and formulate appropriate policy (Quist and Hubert 2004).

The Brook Trout *Salvelinus fontinalis* is the only stream-dwelling trout native to Pennsylvania and much of the eastern United States; however, its historic range has been drastically reduced largely due to habitat degradation, as well as introduction and invasion of nonnative salmonids (EBTJV 2011). The existence of nonnative salmonids in many waterways can be attributed primarily to stocking practices initiated during the 19th century (Behnke 2002). Current Brook Trout distributions in Pennsylvania are often reduced to small, relatively high-elevation headwater streams, but were historically distributed throughout

a much greater proportion of Pennsylvania's waters (Stauffer et al. 2016). Similar distribution patterns were described in the southern Appalachian Mountains (Larson and Moore 1985; Flebbe 1994).

Big Spring Creek, in south central Pennsylvania, is a moderate-sized, low-gradient, alkaline spring stream, which was historically renowned for supporting a high-quality wild Brook Trout fishery (Cooper and Scherer 1967). Prior to the 1930s, Big Spring Creek was reported to support a dense population of Brook Trout along most of its length; however, by the 1950s, Brook Trout were reported to be more localized and no longer widespread in the watershed. Throughout modern history, numerous impacts associated with agricultural activity and the operation of mill dams, clay mines, and trout hatcheries, have altered the chemical and physical characteristics of Big Spring Creek, resulting in severely degraded fish habitat conditions in some segments of the creek (Greene 2002).

Big Spring Creek is unique relative to other wild Brook Trout fisheries in Pennsylvania, as it is one of few low-elevation, low-gradient limestone spring streams that support a robust wild Brook Trout population. Nearly all other spring streams in Pennsylvania support naturalized populations of Brown Trout *Salmo trutta*, or to a lesser extent Rainbow Trout *Oncorhynchus mykiss*. Fertility attributable to numerous limestone springs in the Big Spring Creek watershed creates conditions conducive to produce relatively large (30 – 45 cm TL) Brook Trout, unlike most Pennsylvania populations that occur in small, infertile, headwater streams. Currently, sympatric populations of Brook Trout, Rainbow Trout, and Brown Trout reside in Big Spring Creek; however, the trout species composition varies markedly along its length. Prior to 2010, the wild trout distribution transitioned from a wild Brook Trout dominated system in the upstream-most portion of the creek to a wild Rainbow Trout dominated system immediately downstream. Low densities of Brown Trout occurred in Big Spring Creek upstream from the Laughlin Mill Dam located at RKM 2.1, which precluded upstream passage of that species from a robust population located downstream from the dam. Additionally, Brown Trout reproduction was poor upstream from the dam.

To bolster the Brook Trout population residing in Big Spring Creek, the Pennsylvania Fish and Boat Commission, along with multiple project partners, implemented two habitat enhancement projects in

2010 and 2013 with the intent of optimizing the Brook Trout fishery while recognizing the local social and economic importance of the naturalized nonnative Rainbow Trout fishery. As such, these projects were designed to preferentially favor Brook Trout. In doing so, these projects follow the agency's management strategy applied to this portion of the stream, which is to manage the stream as a sympatric salmonid fishery, but to manage preferentially for Brook Trout without removal of nonnative salmonids. The response of the trout populations to habitat enhancement was used to measure project success towards meeting agency objectives. The purpose of this paper is to describe that response and discuss implications in the context of native and nonnative sportfish management.

Methods

Big Spring Creek is an 8.2-km long tributary to the Conodoguinet Creek in the middle Susquehanna watershed in south central Pennsylvania (Figure 1). The stream forms from springs that originate from karst limestone formations in the vicinity of Newville, Pennsylvania, at 155 m msl. Big Spring Creek has a mean annual discharge of 0.84 m³/s near the spring source (<https://waterdata.usgs.gov/pa/nwis>) and is characteristic of a moderate-sized, low-gradient, alkaline (mean alkalinity as CaCO₃ of 192 mg/L) spring stream.

Habitat enhancement efforts were focused at the upstream portion of Big Spring Creek managed with catch-and-release, fly-fishing only angling regulations. Upper Big Spring Creek flows through hardwood dominated riparian zones adjacent to agricultural and rural residential development. This portion of the stream extends 2.5 km downstream from the spring source and the treatment reach encompassed approximately 1.2 km of linear stream length. Habitat enhancement projects were constructed in two phases in 2010 and 2013 and formed a continuous treatment reach once completed; henceforth referred to as phase 1 and phase 2, respectively (Figure 2).

Projects were designed to model the characteristics of an upstream segment of the stream; henceforth referred to as the model reach. The model reach was the only un-impounded segment of Big Spring Creek where Brook Trout biomass and abundance exceeded that of Rainbow Trout and Brown Trout. This stream reach was characterized by extensive

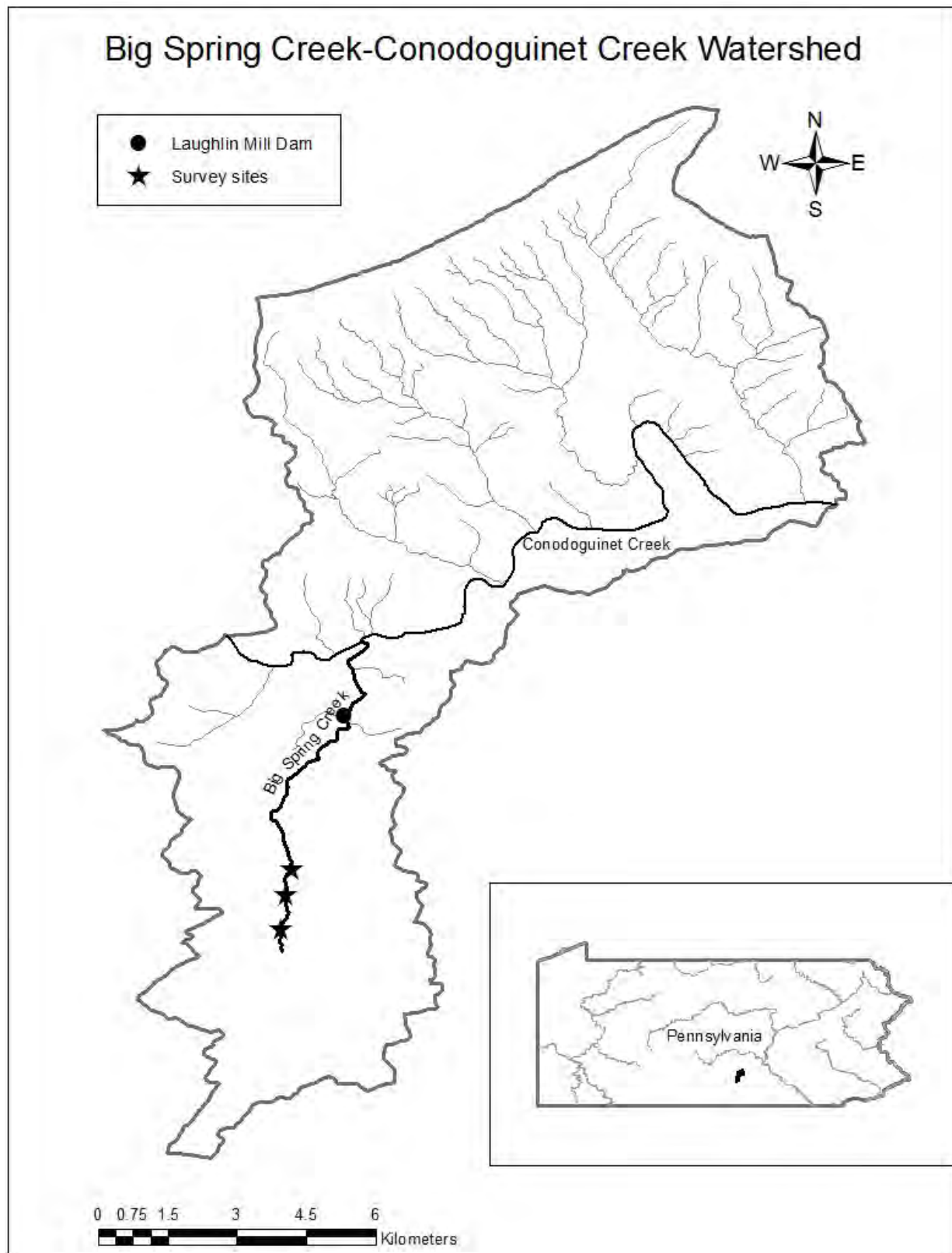


Figure 1. Big Spring Creek watershed, Pennsylvania, showing the three survey sites located at RKM 7.68 (model), 6.87 (phase 1), and 6.24 (phase 2).



Figure 2. Pre- and posttreatment photographs of a portion of the phase 2 habitat enhancement. Photographs provided by Gleim Environmental Group.

physical habitat enhancements constructed in the early 1990s in the form of log-faced deflectors, overhead cover deflectors, single and multi-log vanes, and a channel block (Lutz 2007). These structures narrowed the channel, deepened the water depth, and increased cover for trout.

Furthermore, the physical habitat attributes of the model reach including water column velocity, water depth, substrate particle size, and percent fish cover were within the optimal range of values for adult, juvenile, and spawning Brook Trout reported in select published literature and fisheries agency reports (Reiser and Wesche 1977; Raleigh 1982; Witzel and MacCrimmon 1983; Raleigh et al. 1984; Simonson et al. 1993; Pert and Erman 1994; Baker and Coon 1995; SRBC 1998; Stoneman and Jones 2000; PCWA 2010). However, given the similarities in physical habitat requirements and preferences of stream-dwelling

Brook Trout and Rainbow Trout, many of these attributes were also within the optimal range of values for Rainbow Trout. Despite similarities in habitat suitability, the treatment reaches were designed to be within the optimal range of values for adult, juvenile, and spawning Brook Trout for water column velocity, water depth, substrate particle size, and percent fish cover; however, when possible these attributes were refined to be at the extremes of the optimal range or in the suboptimal range for Rainbow Trout. In general, the habitat enhancement projects narrowed the channel with a series of riparian wetland shelves, deepened the water and slowed the velocity with a series of log water staging devices, and increased the amount and quality of fish cover through a series of undercut banks coupled with selective placement of large woody debris and boulders in the stream channel. A detailed account of the physical habitat attributes and

parameters used to design the projects is provided by Kuhn and McGarrell (2012).

Electrofishing surveys designed to evaluate the response of the trout populations to the habitat enhancement projects were conducted annually during summer from 2008 to 2016 at survey sites located in both the phase 1 and phase 2 project reaches, as well as the model reach. The phase 1, phase 2, and model reach survey sites were 691 m, 325 m, and 300 m long, respectively. The phase 1 electrofishing survey site comprised the entire length of the 2010 project, while the phase 2 electrofishing survey site located at the downstream portion of the 2013 project comprised 53% of that treatment.

Salmonid populations were sampled with towed boat electrofishing techniques utilizing pulsed direct current. Captured trout were measured, recorded in 25-mm TL groups and given an identifying upper caudal fin mark during the initial electrofishing pass to facilitate a mark-recapture population estimate. Trout population estimates were determined using the Chapman modification of the Petersen estimator or M+C-R when R was less than three (Ricker 1975). Simple linear regression was used to identify biomass and abundance trends through time. These analyses were performed using R (R Core Team 2017), and an alpha of 0.05 was the decision criterion used to determine significance. Statewide average weights calculated for each 25-mm length group utilizing the Pennsylvania Fish and Boat Commission's agency database were used to generate the biomass (kg/ha) estimates. The surface areas used in the biomass calculations were held constant for pre- and post-construction time periods to reflect changes in surface area that resulted from the habitat projects. Additionally, annual variations in stream width associated with discharge were minor due to the consistent nature of Big Spring Creek's discharge pattern. Length-frequency analysis was used to estimate the minimum 25-mm length groups of age-1 and older trout; henceforth referred to as adult trout.

Results

Brown Trout accounted for less than 1% of all captured trout and were consequently excluded from all analyses. Brook Trout greater than or equal 150 mm and Rainbow Trout greater than or equal to 200 mm were classified as adults. Adult Brook Trout

and Rainbow Trout biomass and abundance varied substantially both spatially and temporally from 2008 to 2016 (Figure 3).

Adult Brook Trout biomass and abundance in the model reach site ranged from 33.40 kg/ha in 2016 to 86.47 kg/ha in 2012 and 261 fish/km in 2008 to 836 fish/km in 2012, respectively. No significant linear trends were observed for Brook Trout biomass ($r=-0.129$, $P=0.782$) or abundance ($r=0.249$, $P=0.591$). The proportion of estimated adult trout abundance comprised of adult Brook Trout ranged from 62% in 2016 to 83% in both 2009 and 2015. Adult Rainbow Trout biomass and abundance in the model reach site ranged from 21.87 kg/ha in 2008 to 62.66 kg/ha in 2012 and 90 fish/km in 2008 to 265 fish/km in 2012, respectively. No significant linear trends were observed for Rainbow Trout biomass ($r=0.196$, $P=0.674$) or abundance ($r=0.571$, $P=0.180$). The proportion of estimated adult trout abundance comprised of adult Rainbow Trout ranged from 17% in both 2009 and 2015 to 38% in 2016.

Adult Brook Trout biomass and abundance in the phase 1 site ranged from 1.97 kg/ha in 2008 to 57.39 kg/ha in 2016 and 15 fish/km in 2008 to 660 fish/km in 2015, respectively. Significant positive linear trends were detected for Brook Trout biomass ($r=0.951$, $P<0.001$) and abundance ($r=0.962$, $P<0.001$). The proportion of estimated adult trout abundance comprised of adult Brook Trout ranged from 12% in 2008 to 69% in 2016. Adult Rainbow Trout biomass and abundance in the phase 1 site ranged from 15.27 kg/ha in 2008 to 158.39 kg/ha in 2012 and 112 fish/km in 2008 to 678 fish/km in 2012, respectively. No significant linear trends were observed for Rainbow Trout biomass ($r=0.402$, $P=0.371$) or abundance ($r=0.416$, $P=0.353$). The proportion of estimated adult trout abundance comprised of adult Rainbow Trout ranged from 31% in 2016 to 88% in 2008.

Adult Brook Trout biomass and abundance in the phase 2 site ranged from 1.99 kg/ha in 2012 to 37.94 kg/ha in 2016 and 25 fish/km in 2012 to 552 fish/km in 2016, respectively. Significant positive linear trends were detected for Brook Trout biomass ($r=0.975$, $P=0.005$) and abundance ($r=0.913$, $P=0.030$). The proportion of estimated adult trout abundance comprised of adult Brook Trout ranged from 9% in 2012 to 87% in 2016. Adult Rainbow Trout biomass and abundance in the phase 2 site ranged from 11.16 kg/ha in 2011 to 63.27 kg/ha in 2014 and 83 fish/

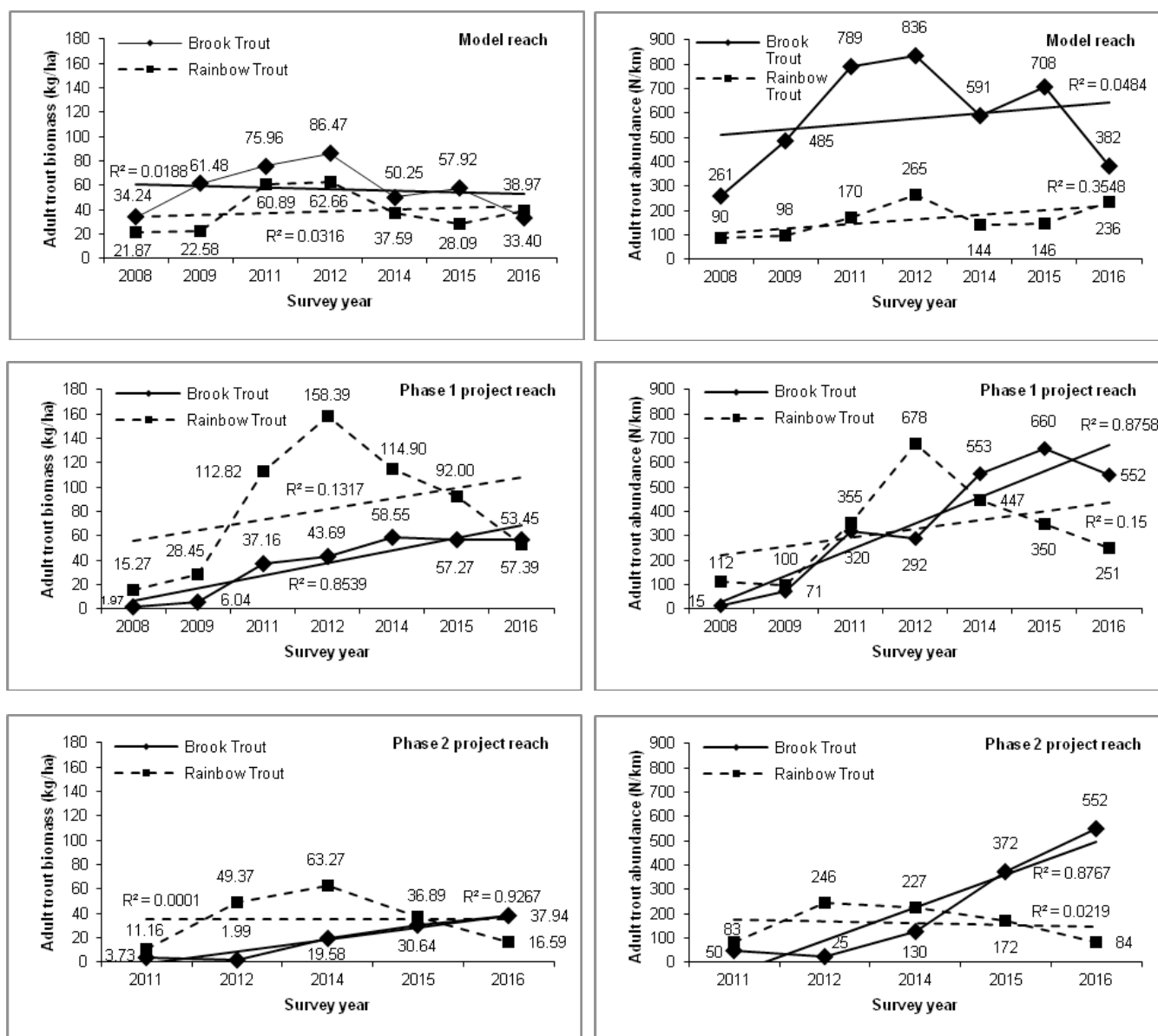


Figure 3. Estimated adult Brook Trout and Rainbow Trout biomass and abundance (number [N] per kilometer) at Big Spring Creek, Pennsylvania, 2008–2016. Linear regression lines for Brook Trout (solid line with no symbols) and Rainbow Trout (dashed line with no symbols) are shown with R^2 values. The phase 1 project was constructed in 2010 and the phase 2 project in 2013.

km in 2011 to 246 fish/km in 2012, respectively. No significant linear trends were observed for Rainbow Trout biomass ($r=0.048$, $P=0.939$) or abundance ($r=-0.119$, $P=0.848$). The proportion of estimated adult trout abundance comprised of adult Rainbow Trout ranged from 13% in 2016 to 91% in 2012.

In summary, Brook Trout and Rainbow Trout biomass and abundance in the model reach have varied during the period of record, and no trend in this variation was observed through time. Brook Trout

residing in the phase 1 project reach have substantially and steadily increased from pretreatment population levels. Likewise, Rainbow Trout population levels have also increased during this same period. However, Rainbow Trout biomass and abundance have declined in recent years following a substantial increase in population size attributable to large year classes one and two years posttreatment. Similar population trends were documented in the phase 2 project reach.

Discussion

Despite improvements documented in the Brook Trout population one-year posttreatment, this initial proliferation of Rainbow Trout in the phase 1 project reach prompted agency biologists to update and further define the management goal and objectives applied to Big Spring Creek. The management strategy applied to the upstream portion of the creek was modified to treat the stream as a sympatric trout fishery, but to manage the fishery preferentially for Brook Trout without removal of nonnative salmonids. Objectives pertaining to this management strategy were to achieve and maintain an adult salmonid biomass of greater than or equal to 90 kg/ha and species abundance composition of at least 70% Brook Trout to 30% Rainbow Trout within 7 years of completed habitat enhancement projects. These criteria were determined using median values at long-term monitoring sites located within the model reach where Brook Trout were thriving and within the phase 1 project reach where Rainbow Trout were dominant.

This approach considered ecological, social, and economic factors pertaining to the Big Spring Creek trout fishery while placing an emphasis on native Brook Trout conservation. In doing so, it also recognized the popularity of the existing Rainbow Trout fishery. Perhaps the greatest disparity between Brook Trout and nonnative salmonid fisheries occurs with the perceived versus actual differences in their recreational values. As such, replacement of a portion of Brook Trout fishery with nonnative Rainbow Trout may have only a slight negative effect or even a positive effect on the recreational fishery. In the western United States, angler preference among salmonids may be minor or nonexistent and evidence suggests that many anglers prefer nonnative salmonids over natives due to their perceived superior sporting qualities (i.e. fighting ability, jumping ability, etc.) and their larger maximum length (Quist and Hubert 2004). Some anglers and fishing guides who frequently fish Big Spring Creek have reported a preference to agency biologists for Rainbow Trout compared to Brook Trout due to their larger body size (personal communications).

The economic issues associated with Brook Trout restoration are directly related to the previously described social issue regarding preference of some anglers for nonnative salmonids. In the western United States, Quist and Hubert (2004) asserted that the net

economic benefit linked to replacement of Cutthroat Trout *Oncorhynchus clarkii* by nonnative salmonids depended on the values society attributed to a particular species, and this concept is likely applicable to native species management in Pennsylvania as well. Nonnative salmonids are likely to be considered ecological and social surrogates for natives in most cases (Quist and Hubert 2004), thus their presence is likely to result in no net economic change to the value of the fishery.

Furthermore, removal through either active or passive means would likely be ineffective, costly, and opposed by the public. To eliminate or reduce encroachment of nonnative fish species, natural resource agencies throughout the United States have utilized a variety of techniques including electrofishing removal, chemical renovation, and isolation management with barriers to prevent reinvasion of nonnative species. These efforts have resulted in varied success, are labor intensive over a relatively long-time period, and are often quite costly (Moore et al. 1983; Larson and Moore 1985; Thompson and Rahel 1996; Kulp and Moore 2000; Quist and Hubert 2004; Avenetti et al. 2006; Meyer et al. 2006; Carmona-Catot et al. 2010).

Variation observed in the populations of Brook Trout and Rainbow Trout at the model reach with no significant trend detected suggests that the significant positive trends of Brook Trout populations observed in the project reaches were independent from environmental effects and likely attributable to habitat enhancement efforts. Despite achievement of and progress towards project objectives, ecological factors may preclude long-term sustainability of this trend. Due to similarities in habitat suitability among stream-dwelling salmonids, the population trends documented here were somewhat unexpected. Weigel and Sorensen (2001) determined that physical habitat characteristic of a Minnesota spring stream did not explain all the variation in densities and distribution of sympatric trout populations. Based on the ecology of sympatric Brook Trout and Rainbow Trout populations, some reduction of Rainbow Trout will likely be necessary to meet agency objectives. As previously stated, harvest of trout is prohibited in the study area; however, from 1976 to 1995 this stream section was managed with special regulations that allowed for the harvest of two trout per day greater than or equal to 381 mm. That regulation was

intended to encourage harvest of Rainbow Trout and Brown Trout and to protect most of the Brook Trout population (Greene 2002). Likewise, in 2012, agency biologists proposed a selective harvest regulation that would continue to restrict gear to fly-fishing tackle, but permit the harvest of Rainbow Trout and Brown Trout while no-harvest regulations for Brook Trout would remain in effect. This regulation was crafted to provide fly-fishing anglers the opportunity to harvest fish and further aide in maintaining and promoting a wild trout community dominated by Brook Trout; however, the proposal was met with strong public opposition and subsequently not implemented. Angler reluctance to harvest nonnative trout in this instance may preclude long-term achievement of management objectives. This situation illustrates the importance of the value society may place on nonnative fisheries. To effectively advocate for native species management efforts such as those described here, biologists must bridge the gap among science, societal values, and economics so that anglers and policy makers can make informed decisions regarding native species conservation techniques such as selective harvest of nonnative species.

Acknowledgements

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Brook Trout Response to Strategic Wood Additions in the East Branch Nulhegan River Watershed, Vermont

Jud F. Kratzer¹, Joseph A. Norton²

¹Vermont Fish and Wildlife Department, 374 Emerson Falls Road, Suite 4, St. Johnsbury, VT 05819, USA

²Trout Unlimited, 97 Main Street, Suite 105, Lancaster, NH 03584, USA

Abstract—Log driving and other poor logging practices have severely degraded fish habitat in many rivers in northern New England. The East Branch Nulhegan River in northeastern Vermont is one of many rivers in this region that continues to suffer from past damage caused by multiple rounds of clear cutting, construction and operation of seven log driving dams along its 27-km length, and removal of large boulders and large wood from the channel. Much of the East Branch remains wide and shallow with little cover for salmonids. In 2012 and 2013, 107 large woody material structures were constructed using chain saw and grip hoist in the East Branch and two of its tributaries. Twelve pairs of control and treatment sites have been electrofished annually starting 2 years prior to wood additions in the treatment sites. By the third year of posttreatment sampling, the average increase in Brook Trout *Salvelinus fontinalis* biomass at treatment sites was approximately 150%. Mean Brook Trout biomass also increased at control sites, and it is possible that this increase is related to large wood additions in treatment sites and other reaches.

Introduction

Log driving and other poor logging practices have severely degraded fish habitat in many rivers in northern New England. The 27-km long East Branch Nulhegan River in northeastern Vermont is one of many rivers in this region that continues to suffer from past damage from multiple rounds of clear cutting, construction and operation of seven log driving dams, and removal of large boulders and large wood from the channel to aid in driving logs to downstream mills. The over-widened, shallow reaches of the East Branch and its tributaries are generally lacking in cover for the native salmonid, Brook Trout *Salvelinus fontinalis*.

Large woody material is especially lacking in the East Branch and its tributaries. Kratzer and Warren (2013) found that water temperatures, depth, and large wood were the three most important factors limiting Brook Trout biomass in northeastern Vermont streams. In response to this study, Trout Unlimited and the Vermont Fish and Wildlife Department began what we call “strategic wood additions” in northeastern Vermont streams in 2012. Large wood structures were constructed at 107 locations in the East Branch and tributaries in 2012 and 2013. These structures, each consisting of one to ten pieces of large wood (greater than 10 cm diameter), were constructed by using chain saws to fell riparian trees and a grip hoist to position

them in the stream channel (Figure 1). Each structure was anchored in place by locking some or all logs between stumps or standing trees on the bank. To date, we only know of seven structures in the East Branch watershed that have been completely lost from their original locations, but most of this wood was deposited downstream within the channel or in the floodplain. The East Branch has deposited the logs from several structures on the upstream end of a large island, and this log jam has caused increased flow on one side of the island at the expense of flow on the other side (Figure 2).

Several studies have documented increases in salmonid abundance in response to additions of large woody material (Jones et al. 2014; Roni et al. 2014; Pierce et al. 2015; Louhi et al. 2016). Until recently most wood addition practitioners used either heavy equipment to precisely place and anchor large wood in the stream channel, or they used chainsaws to “chop-and-drop” riparian trees with no additional efforts made to re-position the felled trees. The strategic wood addition methods used in the East Branch watershed, including the use of a grip hoist to position trees and lock them in place, are under-represented in the literature. The purpose of this study was to determine whether strategic wood additions contributed to increased Brook Trout biomass in the East Branch Nulhegan River and its tributaries.

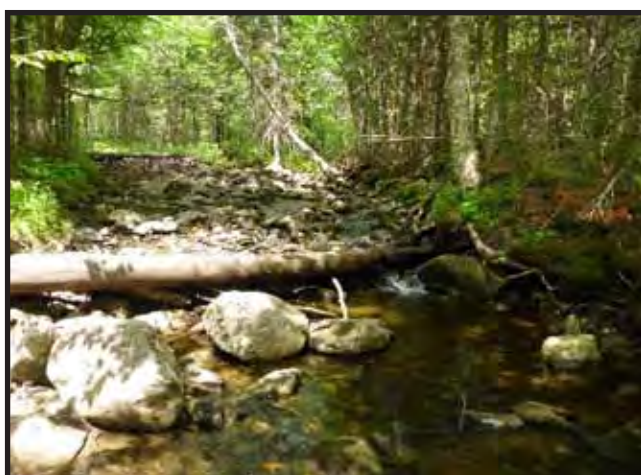


Figure 1. Photos of grip hoist in operation (top left), log locked in place between two standing trees on the stream bank (top right), and completed strategic wood addition structures on the East Branch Nulhegan River (middle row) and Fisher Brook (bottom row).

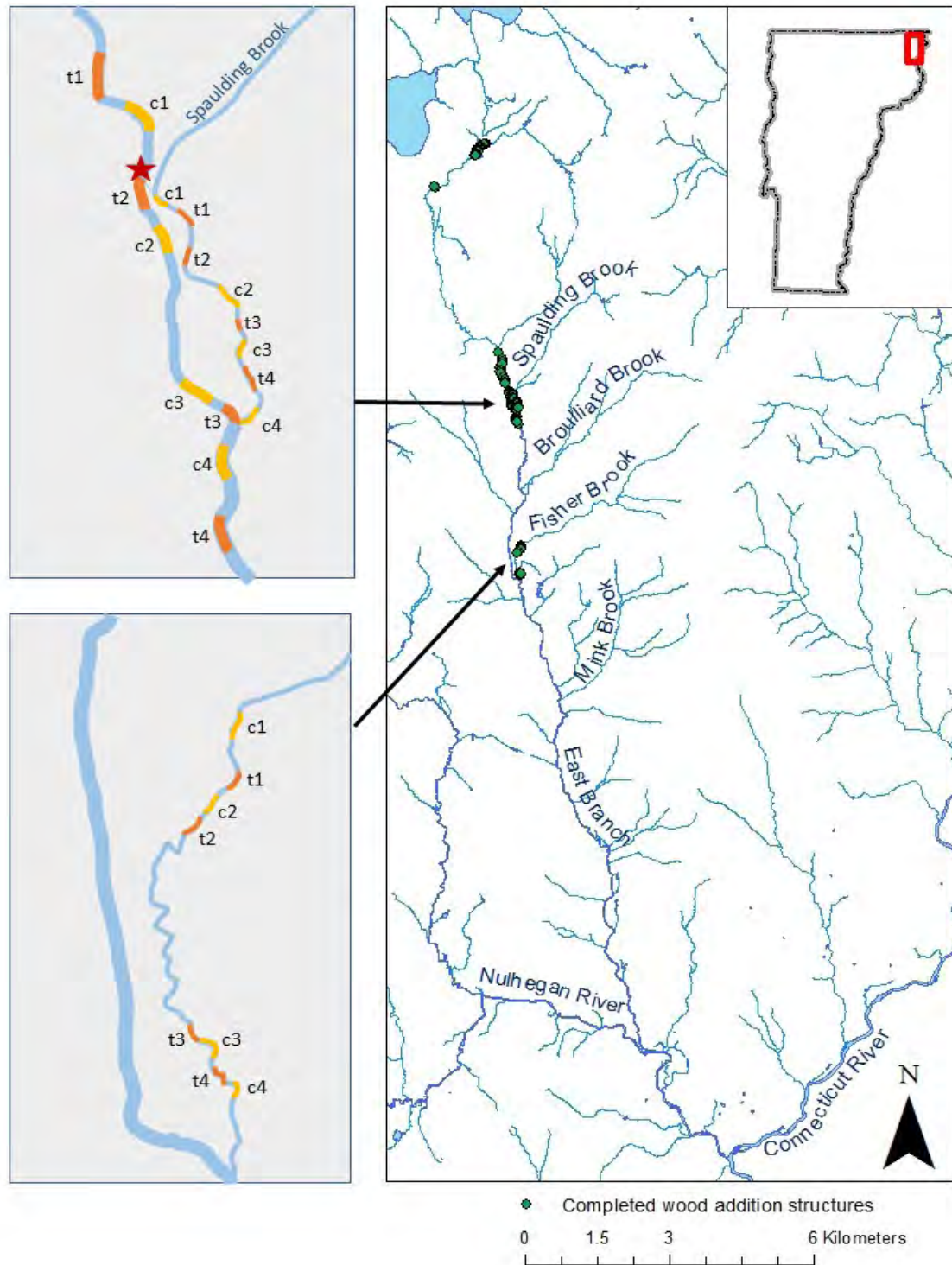


Figure 2. Map of the East Branch Nulhegan River watershed showing locations of large wood structures and control and treatment sampling sites. Red star indicates the location where a large wood jam, formed after the 2013 sampling, now diverts a large proportion of the East Branch flow to the east side of the island.

Methods

All sampling sites included in this study were in the East Branch Nulhegan River and its tributaries (Figure 2). Nearly the entire East Branch watershed is owned by a large private timber company, which is actively managing the land for forest products. Conservation and public access easements preclude logging in riparian areas and guarantee public access.

In 2012, we selected 12 pairs of control and treatment sites, with four pairs in each of three streams: East Branch, Fisher Brook, and Spaulding Brook (Table 1). Control and treatment sites were generally separated by at least two riffles and were selected as pairs for a before-after-control-impact analysis, which will occur in the future. In 2012 and 2013, crews constructed 107 large wood structures consisting of one to ten pieces of large wood in the East Branch, Fisher Brook, and Spaulding Brook. We built forty-three of these structures at treatment sites after sampling was completed in 2013. We also selected three additional “control brooks” sites. Distance and/or culverts that inhibit the upstream movement of Brook Trout isolate these control brooks sites from reaches where wood was added. Sampling sites ranged in length from 24 to 65 m, with an average of 41 m.

We sampled Brook Trout populations at the 27 sampling sites during low flow conditions each July from 2012 to 2016. We used one or two backpack electrofishing units to capture Brook Trout. We often needed two backpack units to effectively draw fish

Table 1. Characteristics of the 12 pairs of control and treatment sites on East Branch, Fisher Brook, and Spaulding Brook and of the three control brooks sites on Broulliard, Fisher, and Mink Brooks. Bankfull width and slope values for East Branch, Fisher Brook, and Spaulding Brook are the range of values for all sites in those streams.

Stream	Stream order	Bankfull width (m)	Slope
Broulliard	2 nd	10	4%
East Branch	3 rd	9-15	1-3%
Fisher	2 nd	5-8	0-3%
Mink	2 nd	5	4%
Spaulding	2 nd	5-9	1-2%

from large wood structures. We used multiple pass depletion (Carle and Strub 1978), without block nets, to estimate Brook Trout abundance.

Results

Mean Brook Trout biomass increased at treatment sites following the addition of large woody material in 2013. The increase in Brook Trout biomass at treatment sites was evident for all sizes of Brook Trout combined and for large Brook Trout (greater than 152 mm TL, Figure 2). Mean Brook Trout biomass for all sizes slightly decreased at control sites the first year after wood addition, but increased in the second and third years. Mean Brook Trout biomass at the control brooks sites was relatively high from 2010 to 2012 before dropping in 2013, after which it remained relatively low. This decline in biomass at the control brooks sites was more pronounced for large Brook Trout. The cause of this decline is unknown, but it may partially be a result of small sample size (n=3).

Brook Trout biomass increased at all treatment sites, at all but two control sites, and at one control brook site following wood addition (Table 2). The observed increases in Brook Trout biomass were at least partially related to changes in large wood density at the treatment sites and other reaches. In general, treatment sites that experienced the greatest percent increase in Brook Trout biomass were those sites that had the highest wood densities in 2016. It is possible that the increases observed at most of the control sites were also partially related to increases in wood density. While wood was not added to the control sites, and only one log was observed to have drifted into a control site from upstream, Brook Trout were free to move between control sites and reaches where wood was added. In contrast, Brook Trout biomass decreased at two of the three control brooks sites, which are isolated by distance and/or impassable culverts from wood addition sites.

Discussion

The addition of large woody material clearly contributed to increased Brook Trout biomass at treatment sites. This result was expected, in part because of Kratzer and Warren (2013), who found that water temperature and large woody material were the two most important limiting factors for Brook Trout streams in northeastern Vermont. Before we added wood, temperature monitoring in the East

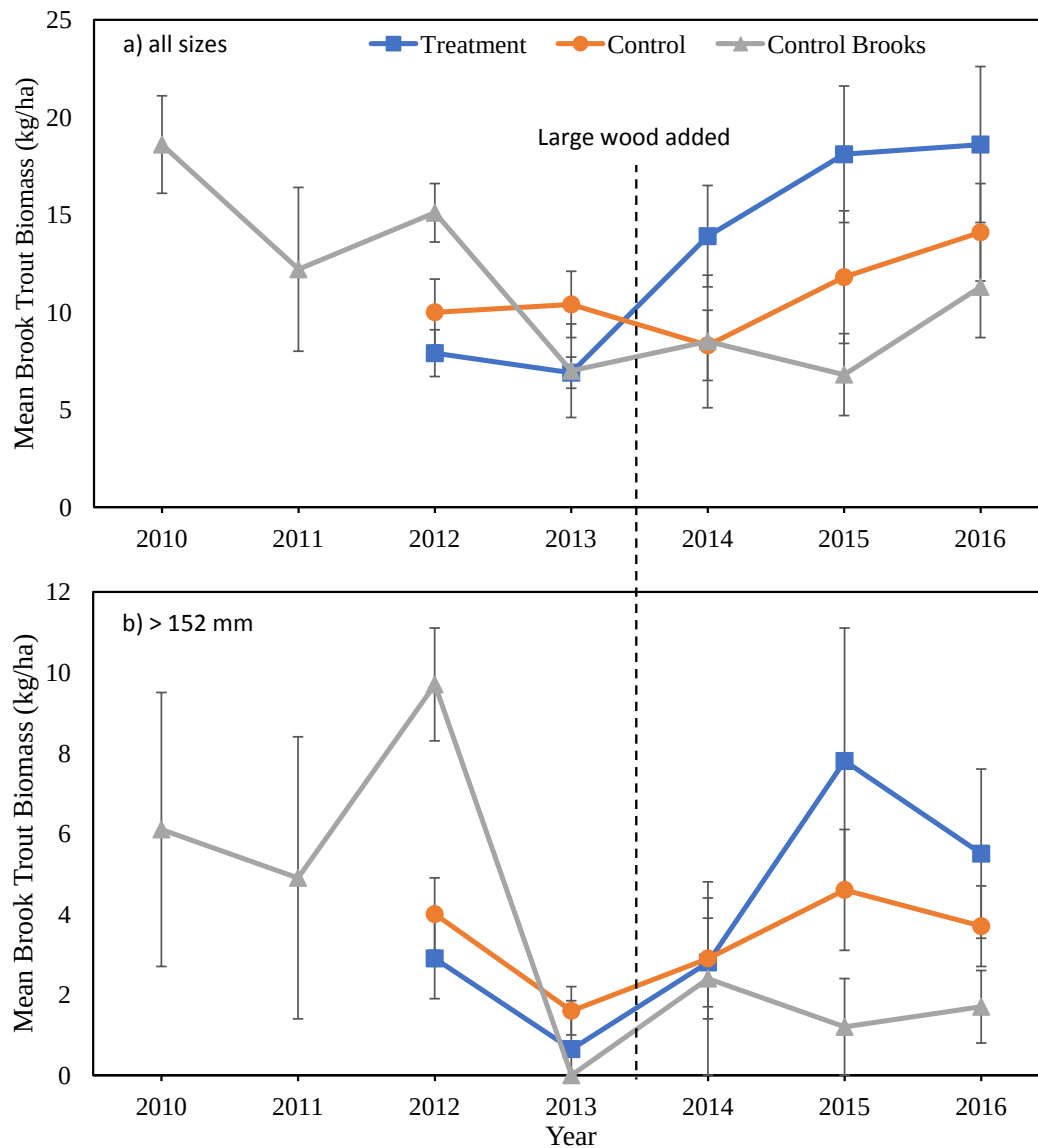


Figure 2. Mean biomass of a) all sizes of Brook Trout and b) Brook Trout greater than 152 mm TL at the 12 treatment and 12 control sites and at three sites that are in the same watershed but isolated by distance and/or impassable culverts (control brooks). Large woody material was added to treatment sites (and other reaches not included in this study) following the 2013 sampling. Error bars display mean $\pm$ one standard error.

Branch and Fisher Brook had demonstrated that water temperatures were suitable for Brook Trout in these streams, and habitat assessments demonstrated that large woody material was generally lacking (Vermont Fish and Wildlife Department, unpublished data). Increased Brook Trout biomass at treatment sites was also expected because of several studies that have demonstrated the positive relationship between large

wood and salmonid populations (Jones et al. 2014; Roni et al. 2014; Pierce et al. 2015; Louhi et al. 2016).

Brook Trout biomass increased by at least 67% at eight of the twelve treatment sites after large wood was added. The relatively small (20%) increase at one site, East Branch treatment 1, was likely the result of low wood density. Kratzer and Warren (2013) found that Brook Trout biomass was positively correlated

Table 2. Mean Brook Trout biomass at the 12 treatment and 12 control sites and at the three control brooks sites during 2 years before (2012-13) and 2 years after (2015-16) large woody material was added to treatment sites and other reaches, sorted in order of decreasing percent change. Wood density is the number of pieces of large wood per hectare of wetted area in July 2016. Flow change is a qualitative assessment of changes in discharge resulting from large woody material additions, which occurred after sampling in 2013.

Stream	Site	Mean biomass (kg/ha)			Wood density (number/ha)	Flow change
		2012-13	2015-16	% Change		
Spaulding	treatment 4	9.9	49.4	399%	893	increase
Spaulding	treatment 3	7.3	36.3	397%	1557	increase
East Branch	treatment 4	3.2	13.8	331%	523	none
Fisher	treatment 2	5.2	18.8	262%	1416	none
Fisher	treatment 3	2.5	8.8	252%	1370	none
Spaulding	treatment 2	4.7	11.8	151%	551	none
Fisher	control 3	13.7	29.4	115%	1260	none
Spaulding	treatment 1	7.2	15.2	111%	1037	decrease
Spaulding	control 4	5.9	10.6	80%	342	increase
Spaulding	control 1	22.1	38.6	75%	1165	increase
Fisher	treatment 4	9.4	15.7	67%	1557	none
Fisher	control 2	7.5	10.3	37%	460	none
Fisher	control 1	4.3	5.7	33%	466	none
Spaulding	control 3	10.5	13.8	31%	365	increase
Fisher	treatment 1	10	12.9	29%	1369	none
East Branch	control 3	9.2	11.6	26%	271	decrease
Fisher	control brook	8.7	10.9	25%	365	none
East Branch	treatment 1	10.5	12.6	20%	252	none
East Branch	control 4	4.7	5.5	17%	94	none
Spaulding	control 2	8.8	9.9	13%	205	increase
East Branch	treatment 2	7.9	8.8	11%	495	decrease
East Branch	control 1	11.3	12.2	8%	137	none
East Branch	treatment 3	10.8	11.2	4%	917	decrease
East Branch	control 2	7.2	7.2	0%	289	decrease
Mink	control brook	14.9	11.8	-21%	563	none
Fisher	control 4	17	9.4	-45%	718	none
Broulliard	control brook	9.5	4.5	-53%	81	none

with wood density only when the latter exceeded approximately 200 pieces per hectare. Wood density at East Branch treatment 1 was only 252 pieces per hectare in 2016. The next lowest wood density at a treatment site was 495 pieces per hectare.

Changes in flow patterns were an unforeseen confounding factor. Between the 2013 and 2014 sampling events, the East Branch dislodged a large

wood structure from a reach that was not included in this study and deposited it on the upstream end of the island where Spaulding Brook meets the East Branch valley bottom (Figure 2). Since that time, this new log jam has been functioning to divert a large proportion of the East Branch's discharge into the channel that had formerly been dominated by Spaulding Brook discharge. The result is that two pairs of East Branch

control and treatment sites are now receiving reduced discharge during both high and low flows, while most of the Spaulding Brook treatment and control sites are now receiving increased discharge. When we first discovered this unplanned log jam, we contemplated removing it to restore pre-treatment flow patterns, but we decided against it because we expected that, within each pair, control and treatment sites would receive the same discharge. This expectation was met for some pairs, but not for others. Additional channel braiding induced by 2013 wood additions caused three treatment sites (East Branch 3, Spaulding 1, and Spaulding 2) to receive lower discharges than their corresponding control sites. Despite the relatively low discharge, Brook Trout biomass more than doubled at both of these Spaulding Brook treatment sites, presumably because of improved habitat. Of the four treatment sites that experienced a less than 30% increase in Brook Trout biomass, two of them were the East Branch sites (2 and 3) that received reduced discharge after 2013. Increased discharge may have contributed to increased Brook Trout biomass at all Spaulding Brook control sites.

While changes in flow patterns were an unforeseen consequence of wood additions, they may ultimately provide an unforeseen benefit to Brook Trout. Braiding on the east side of the island created approximately 570 m² of new Brook Trout habitat. When we electrofished this new habitat in July 2016, we found a total biomass of approximately 1 kg of Brook Trout, or 18.3 kg/ha, which was comparable to some of the higher quality treatment sites (Table 2). This result would not be unexpected by the many river restoration practitioners that are purposely trying to restore braiding in rivers and reinvigorate abandoned side channels (Bellmore et al. 2013).

Many researchers, fisheries managers, and stakeholders have wondered whether fish habitat improvement efforts actually increase the overall fish population or whether areas of improved habitat simply concentrate fish that otherwise would have been widely dispersed across areas of lower quality habitat. Gowan and Fausch (1996a) concluded that habitat improvement can benefit salmonid populations beyond treated reaches if fish dispersal is largely unimpeded within the drainage. A stream naturally has areas of varying habitat quality, which can be expected to support varying abundances of trout. When habitat is improved in portions of a stream, the improved areas

can support more trout, some of which may be drawn out of areas of poorer quality habitat. Vacated habitats are then able to support additional trout, assuming recruitment is not limiting. The first year (2014) after large wood was added to treatment sites and other reaches, mean Brook Trout biomass increased at treatment sites but decreased at control sites, possibly because some Brook Trout were drawn out of areas of poor habitat to areas of improved habitat. However, in the second and third years following wood addition, mean Brook Trout biomass was higher than pretreatment at both treatment and control sites, which may suggest that increases in large wood are contributing to increased Brook Trout population size in the entire system. Large wood directly benefits adult salmonids by providing cover, but it can also benefit recruitment by retaining gravels, which benefit spawning, and by providing cover for juvenile salmonids (Dolloff and Warren 2003; Whiteway et al. 2010; Roni et al. 2014; Jones et al. 2014). Large wood can also increase food for trout by retaining organic material, which provides food for invertebrates (Benke and Wallace 2003). Increases in recruitment and invertebrate abundances could contribute to increased Brook Trout abundance, even in reaches where wood was not added. Mean biomass at control brooks sites has remained relatively low since 2013 but did increase slightly in 2016. A few more years of data should help to determine whether recent increases at control sites are related to increases in large wood in other reaches or to some other factors that have benefitted regional Brook Trout populations, including those in the control brooks.

One of the purposes for trout habitat improvement efforts in the East Branch Nulhegan River watershed was to improve Brook Trout fishing for anglers. Vermont Fish and Wildlife Department uses the benchmark of 152 mm (6") as the minimum sized Brook Trout that would be of interest to anglers. We found that mean biomass of Brook Trout greater than 152 mm increased at treatment sites following wood addition. In fact, it nearly doubled from 2012 to 2016. It is possible that wood additions are contributing to increased large Brook Trout biomass at control sites, but more years of data will be required to determine whether this is the case. It is likely that the full effect of wood additions on large Brook Trout abundance cannot be demonstrated by this study. While wood addition has contributed to improved Brook Trout

habitat in the study area, adult Brook Trout can be very mobile (Gowan and Fausch 1996b; Adams et al. 2000; Roghair and Dolloff 2005; Petty et al. 2012), and there is nothing to prevent Brook Trout from moving out of the study area and taking up residence in larger, deeper portions of the East Branch, the Nulhegan River, or the Connecticut River. Indeed, one of the goals of wood additions in the Nulhegan River watershed has been to benefit trout populations in the entire upper Connecticut River watershed. Angler harvest could also make it difficult to fully quantify the effect of our strategic wood additions on large Brook Trout.

This study will continue for at least one more year, possibly several more. We electrofished all treatment, control, and control brooks sites in July 2017, but these data were not analyzed in time for inclusion in this manuscript. After the 2017 data are analyzed, we will decide whether to sample again in 2018 before drafting a manuscript for submission to a peer-reviewed journal. We are not likely to sample every year after 2018, but we plan to periodically sample all sites to assess the long-term effects of strategic wood addition on the Brook Trout population.

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Session 8

Disease, Parasites, and the Health of Wild Trout: Should We be Concerned?





Impacts of Nonnative and Invasive Fish Pathogens to North Carolina's Trout Resources and its Managers

J. M. Rash¹, C. F. Ruiz², and S. A. Bullard²

¹North Carolina Wildlife Resources Commission, 654 Fish Hatchery Road, Marion, NC 28752

²Auburn University, School of Fisheries, Aquaculture, and Aquatic Sciences, Southeastern Cooperative Fish Parasite and Disease Laboratory, 203 Swingle Hall, Auburn, AL 36849

Abstract—Three parasitic species that affect salmonid populations were discovered recently in North Carolina: the parasitic copepods (aka “gill lice”) *Salmincola edwardsii* (infections in Brook Trout *Salvelinus fontinalis*; 2014) and *Salmincola californiensis* (infections in Rainbow Trout *Oncorhynchus mykiss*; 2015) as well as the causative agent of whirling disease, *Myxobolus cerebralis* (infections in Rainbow Trout, Brown Trout *Salmo trutta*, and Brook Trout; 2015 and 2016). None of these pathogens previously had been diagnosed taxonomically from North Carolina or from the southeastern United States, but all can exert deleterious population-level effects on salmonids elsewhere in North America and abroad. As such, these pathogens fall within a geographic area where potential biological threats to coldwater resources (including the State's only native salmonid: Brook Trout) are indeterminate. Such knowledge gaps make it difficult to achieve informed decisions on behalf of resource managers. To address this issue, personnel of the North Carolina Wildlife Resources Commission (NCWRC), in collaboration with Auburn University's Southeastern Cooperative Fish Parasite and Disease Laboratory, have obtained and disseminated pathogen-specific information in a step-wise fashion to (1) inform management decisions within the state, (2) disseminate scientific research data to adjacent resource managers likewise concerned with these salmonid pathogens, (3) hasten inter- and intrastate biosecurity measures, and (4) engage in public outreach that targets anglers and other citizens. Although much remains to be investigated, progress has been made as a result of focused studies on disease diagnostics, parasite identification, pathological effects, pathogen temporal and spatial distribution, parasite life cycles, and parasite-host relationships. Research efforts will continue to expand in order to inform management decisions and improve knowledge on the health of North Carolina's salmonids.

Introduction

As with many of the coldwater resources across the United States, trout are a significant biological, cultural, and economic entity for North Carolina. As the State's only native salmonid, the Brook Trout *Salvelinus fontinalis* is a species of conservation importance (NCWRC 2013; 2015) and one that has served as a local fishery for generations of anglers. In addition to self-sustaining populations of Brook Trout, the NCWRC also manages stocked Brook Trout and self-sustaining and stocked Brown Trout *Salmo trutta* and Rainbow Trout *Oncorhynchus mykiss* fisheries through its Public Mountain Trout Waters program. These diverse resources are popular with anglers; approximately 76% of anglers expressed satisfaction with their trout fishing in North Carolina (Responsive Management 2015a). Furthermore, nearly 149,000 anglers fished in Public Mountain Trout Waters

during 2014 and had a total effect of approximately \$383 million to the State's economy (Responsive Management 2015b). Trout also serve as an important agricultural commodity for North Carolina. At approximately \$8.5 million, North Carolina was second to Idaho in total value of trout sold by state in 2016 (USDA 2017).

Unfortunately, there are threats to these important resources that could impact their well-being immediately and into the future. Not limited to North Carolina, fisheries managers in the United States and abroad are dealing with a growing set of challenges associated with introduced species (Li and Moyle 1999; Smith et al. 2008; Thomas et al. 2009; ANSTF 2012; Chapman et al. 2016; Gallardo et al. 2016; Hulme 2017). These species can have varied, but often significant, effects on resident aquatic communities, and when these interactions are proven disruptive, these introduced species are often labeled as nuisances.

Aquatic nuisance species (ANSs) are diverse and include representatives from many animal and plant phyla, including free-living and parasitic species. Free-living ANSs can disrupt food webs and alter substrata within an ecological landscape. For example, the macroalga *Cladophora glomerata* caused nuisance algal blooms in Lake Ontario (Higgins et al. 2008; Kuczynski et al. 2016), and blooms of *Didymosphenia geminata* (an ANS recently discovered in North Carolina; Bowman et al. 2016) have been reported globally (Spaulding and Elwell 2007). While algae like these can alter aquatic ecosystems and inconvenience anglers, exotic Silver Carp *Hypophthalmichthys molitrix* continue to expand their range in North America and can even pose physical risks to boaters (Vetter et al. 2017). Manifestations of ANSs resulting in algal mats or leaping fish resonate with resource users because they interfere with recreation.

Perhaps less obvious to resources users but potentially more impactful to the resource itself, parasitic ANSs can cause infectious diseases among endemic fishes and invertebrates. While free-living ANSs are worrisome because they can have deleterious effects on the behavior and ecology of endemic free-living species, parasitic ANSs are worrisome because they can outright kill endemic free-living species or cause them to be more susceptible to endemic opportunistic pathogens that, under normal conditions, may be benign. In addition, once invasive, the parasitic ANS may persist irrevocably in a particular aquatic ecosystem. Further challenging is that the introduction event is seldom detected immediately nor even years later such that population-level changes are observed before the manager is aware that an invasive pathogen is even present. In this way, ANSs that are parasitic and cause disease can be insidious in aquatic ecosystems.

Recent Discoveries

Three parasite species known to exert deleterious population-level effects on salmonid populations were newly discovered in North Carolina. The parasitic copepods (aka “gill lice”) *Salmincola edwardsii* (infections in Brook Trout) and *Salmincola californiensis* (infections in Rainbow Trout) were identified in 2014 and 2015, respectively (Ruiz et al. 2017). In 2015 and 2016, the causative agent of whirling disease, *Myxobolus cerebralis* (Mc; infections in Rainbow Trout, Brown Trout, and Brook Trout) was

confirmed (Ruiz et al., in review). These observations represent the first time each species has been documented taxonomically in North Carolina and their most southeastern distribution in North America.

Gill lice.—Elsewhere within the United States, *S. edwardsii* and *S. californiensis* are known to infect salmonids of the genera of *Salvelinus* and *Oncorhynchus*, respectively. Taxonomic and molecular analyses of copepods confirmed the identity of both species in the State (Ruiz et al. 2017). To date, *S. californiensis* has been documented on Rainbow Trout only; however, this copepod also has the potential to affect the State’s only Kokanee Salmon *O. nerka* population within Nantahala Reservoir. Unfortunately, little is known about the ecology of copepods of the genus *Salmincola* and their impacts to salmonids in the wild (Black et al. 1983; Amundsen et al. 1997; Hargis et al. 2014).

Given the uncertainty associated with the copepods’ impacts, NCWRC biologists are concerned about what effect these copepods could have on wild trout populations (especially, native Brook Trout populations). As the State’s only native salmonid, Brook Trout are an important ecological and cultural resource: their distribution and populations are reduced from historic levels, and any additional stressor is of concern. Heavy infections of gill lice damage the gill (Ruiz et al. 2017) and can impair fish performance (Conley and Curtis 1993; Amundsen et al. 1997; Alteen 2009; Hargis et al. 2014; Mitro et al. 2014), but low-level infections may be benign (Amundsen et al. 1997). While possible that populations suffering from other stressors (drought, elevated water temperatures, nonnative species) and heavy gill lice infections could experience mortality (Mitro et al. 2014), empirical evidence of these hypothetical interactions is lacking because it is exceedingly difficult to collect and analyze in space and time.

Many additional questions remain. For example, the bulk of the State’s Brook Trout populations are at low density and isolated above 914 m, so what are the characteristics (e.g., rate and mechanisms) associated with the spread of gill lice? Moreover, infection intensity (= the number of gill lice on an individual trout) is proportional to fish body size (Black 1982; Amundsen et al. 1997), so how is this impacting the larger, sexually mature individuals and ultimately, the demography of a population? The examples noted above are a small portion of the questions that could

be examined to allow the NCWRC to understand the short- and long-term impacts of gill lice to salmonids in the state and abroad.

Although gill lice have been documented in selected waters, the full spatial extent of their distribution within North Carolina is unknown. Anglers have been asked to report observations of gill lice during recreational outings, and the NCWRC will continue to examine trout populations in conjunction with ongoing efforts to document the distribution and status of Brook Trout, Rainbow Trout, Brown Trout, and Kokanee Salmon populations in North Carolina.

Whirling disease.—Symptoms of whirling disease are well documented and include “whirling” behavior, spinal cord injury, and brain stem compression (infected fishes cannot achieve equilibrium and swim erratically until exhausted or dead; Elwell et al. 2009). Because the causative agent of whirling disease, *Mc*, infects cartilage, young fish are especially vulnerable. Its life cycle requires an oligochaete (reportedly *Tubifex tubifex*) that is common in sediments of coldwater streams throughout the southeastern United States. As such, infected oligochaetes and the translocation of infected trout are primary risk factors in spreading the pathogen and subsequent disease. To date, *Mc* has been documented in four major river basins in the State (Catawba, French Broad, Watauga, and Yadkin rivers; Ruiz et al., in review).

On July 27, 2015, it was confirmed that Rainbow Trout with clinical signs of whirling disease in the Watauga River, North Carolina, were infected with *Mc* (Ruiz et al., in review). Later that year, additional testing confirmed symptomatic, infected trout in the Elk River, North Carolina. Rainbow Trout from additional sites on the Watauga River and a connected private trout farm were also infected. The NCWRC collected oligochaetes (49 total sediment/mud samples collected in Sept 2015; each with 0–53 oligochaetes; a total of 485 individual oligochaetes subjected to nested PCR) from its Delayed Harvest Trout Waters to screen for the presence of *Mc*. Oligochaetes from Mill Creek and Watauga River were infected. In addition, infected oligochaetes were collected from above the NCWRC’s Marion State Fish Hatchery, and subsequently, this facility has been renovated to incorporate biosecurity measures (UV filters and drum screens) to treat surface waters. In August 2015 (and annually thereafter), trout also were tested from all NCWRC’s trout production facilities; all were negative for infection. Further

results and conclusions from these collections are in preparation to be published in peer-reviewed aquatic animal health and parasitology journals.

It was imperative that the NCWRC understand the scale and scope of *Mc* within the State to inform management decisions. A critical short-term, rapid-response research program was initiated in May 2016 to quickly identify infected trout species and *Mc*-positive streams in North Carolina. This project captured “a snapshot in time” of the immediate status of the exotic, invasive pathogen in North Carolina waters. Using AFS Blue Book protocol (MacConnell and Bartholomew 2014), including the specific confirmatory test (nested PCR) for *Mc*, and including approximately 1,500 trout of 3 species from 36 localities in North Carolina, it was discovered that trout from three major river basins were infected with *Mc*: Brook Trout and Brown Trout from Laurel Creek (Yadkin River Basin), Brown Trout from South Toe Creek (French Broad River Basin), Rainbow Trout from Roaring Creek (French Broad River Basin), and Brown Trout from the Boone Fork (Watauga River Basin). Because of budget and logistical limitations to this study, a relatively small sample size of trout representing each species and each locality were collected and analyzed. However, noteworthy is that the pathogen was still detected, suggesting that the prevalence and intensity of infection by the parasite may be underestimated. In addition, this study comprised a single point in time (no locality was sampled more than once) and served to quickly assess the potential invasive geographic distribution of the pathogen only. Although we know more than we did 2 years ago, the NCWRC will continue to work with Auburn University’s Southeastern Cooperative Fish Parasite and Disease Laboratory researchers to explore the distribution and life history characteristics of *Mc* in North Carolina.

Looking Back

The NCWRC employed a step-wise approach to obtain and disseminate pathogen-specific information. Given the uniqueness of *Mc* and *Salmincola* to North Carolina and the southeastern United States, there was a lack of information to inform management decisions. Initial efforts focused on documenting the geographic distribution and level of host specificity of gill lice infections across all sympatric trout populations (Ruiz et al. 2017). Having this baseline allowed us to seek

information incrementally. For example, Delayed Harvest Trout Waters were sampled in 2015 for *Mc* in advance of these waters receiving stocked fish and a popular catch-and-release period. This information helped to direct outreach efforts (e.g., on-stream signage). The NCWRC then focused on self-sustaining trout populations the following spring to diversify resources examined and increase spatial coverage. Each effort provided important information and informed the next, and future research is planned for both *Mc* and gill lice that will continue to increase our understanding of these ANSs.

While gathering and planning to collect additional information, the NCWRC communicated effectively with management partners. The issues associated with fish pathogens as well as non-infectious ANSs are not confined to administrative boundaries. This is especially clear in North Carolina, where there are numerous waters and watershed boundaries that are shared between states, municipalities, the Great Smoky Mountains National Park, Eastern Band of Cherokee Indian Reservation, and private aquaculture facilities. With this in mind, the NCWRC kept all aware of the progress with managing these newly-discovered coldwater pathogens.

In the case of both *Mc* and gill lice, the NCWRC has utilized angler reports to identify infected trout populations. As such, increasing public awareness has been an important part of the NCWRC's response to these and other ANSs. Press releases and social media posts conveyed relative findings, while a webpage devoted to whirling disease compiled media and provided additional information. On-stream signage was also placed along waterways where *Mc* was documented. Furthermore, the NCWRC has developed an ANS message (Clean, Drain, Dry, and Never Move) that staff can utilize to promote minimal efforts to prevent the spread of ANS.

The NCWRC has also been working to prevent the spread of ANSs via its review of stocking permit applications. In North Carolina, individuals must obtain a permit from the NCWRC to stock any fish in public waters. During the review process, it must be confirmed that trout slated in stockings are free of infections by *Mc* and gill lice (*Salmincola* spp.). To make this determination, the NCWRC has visited suppliers noted by applicants to collect trout for analyses since 2015. Results are communicated with applicants, private facility contacts, and the North

Carolina Department of Agriculture (the State agency with oversight authority over private aquaculture facilities), and if lots are verified to be free of *Mc* and gill lice, the supplier is approved for use.

Although warranted, these efforts are costly. In late August 2015, the NCWRC developed a cost center to better track time and mileage expenditures associated with aquatic nuisance species efforts (primarily activities associated with addressing whirling disease). From 28 August 2015 to 13 July 2017, staff coded 1,247 h and 10,595 mi to that cost center. As one can tell, there was significant amount of effort expended within that approximately 2-year period; much of which was unplanned and all expended at the cost of other activities. Progress has been made as a result of focused studies on parasite species diagnosis, taxonomy and systematics, pathological effects on fishes, pathogen temporal and spatial distribution, and parasite-host relationships. Much remains to be investigated, but research efforts will continue to expand in order to inform management decisions and improve knowledge on the health of North Carolina's salmonids.

Managing in a Parasitic World

Fisheries management is often the manifestation of professional reactions to changing circumstances. Nowhere is this clearer than in the context of adjusting to an exotic ANS while also working to meet biological and socioeconomic goals for fisheries. Can proactive fish health (biosecurity) measures help managers make the best decisions regarding ANSs to reduce impacts to resources and those that manage them? The NCWRC has worked to address newly-discovered fish pathogens and reduce associated biological-knowledge gaps since 2015, and during that time, measures to facilitate such activities were utilized. Yet, there are likely additional opportunities to assist fishery managers in North Carolina and elsewhere. As such, managers should consider the following three focal areas relative to salmonid pathogens within management regimes: research; aquatic animal health planning; and information exchange.

Research.—As noted previously, the NCWRC's stepwise approach has narrowed data gaps regarding *Mc* and gill lice; however, more research is needed. Of paramount importance moving forward are (1) determining the spatial and temporal (seasonal)

distributions of these pathogens in North Carolina trout waters and among its various trout species; and (2) elucidation of the fine details of the life cycles, incidence of disease, and pathological effects of these ANSs on trout. Although much has been published on these various attributes elsewhere, much more evidence is needed for southeastern trout and ecosystems.

Salmincola and *Mc* exemplify salmonid pathogens and are species of focus by the NCWRC, but the need to understand current and potential organisms is consistent across the field of ANS. As pathogens and other ANSs expand their ranges ultimate ramifications to resources are unknown. However, familiarity of ANS identification, ecology, and pathology can provide insight into potential clinical signs of infection, transfer vectors, spatial distribution, information needs, and overall risks of unwanted species; all of which can serve as “warning signs” and foundations to build upon to further knowledge.

Aquatic animal health plans.—All 50 state natural resource agencies make online references to ANSs. While such awareness is encouraging, the amount of species-specific information presented (e.g., handout to formal ANS plan) varied among agencies. Regional differences exist regarding comprehensive, multi-agency aquatic animal health plans. For example, the Northeast Fish Health Committee (NFHC) developed fish health guidelines for member agencies to assist with the importation and transfer of fish, communication, and development of management strategies (NFHC 2015). Although compliance is non-mandatory, the NFHC’s guidelines provide wide-ranging objectives and action items that can be implemented and evaluated. Southeastern fisheries managers face the same challenges of shared waters and fish transfers (including those in the private sector) as their northeastern counterparts but lack a codified fish health plan.

In addition, most fish health guidelines and policies focus on captive, cultured stocks in ponds and raceways. While these are important, there is also the need to develop protocols and best management practices that cater to health monitoring of wild fish populations. If done in conjunction, evaluation of both wild and captive aquatic animals provides a comprehensive approach – something of great importance given each stock’s ability to influence the other.

The topic of health planning brings to mind the old adage regarding an ounce of prevention being better than a dose of the cure. Taking steps to prevent introductions and proliferations of ANSs will result in infrastructure demands (temporal and financial), but those measures of prevention must be weighed against the biological, social, and economic costs of the long-term outcomes of an introduced fish pathogen. Unfortunately, if we revisit the adage, there are no “doses” for the majority of these species once introduced into the wild. Introductions are seldom detected immediately and may be insidious or catastrophic, depending on the interaction among the pathogen, host, and environment (Combes 2001).

Information exchange.—Increased understanding of pathogens and biosecurity measures are important but efforts will not be successful without effective exchange of information regarding all facets of ANSs. The NCWRC will continue to work with anglers (e.g., angler reports and production of educational materials) as their awareness is important to address ANS (Anderson et al. 2014). Furthermore, continued communication with intra- and interstate managers and agencies is critical to ensure the health of wild and captive trout stocks. Salmonid pathogens like *Salmincola* and *Mc* are no longer potential invaders of North Carolina’s trout resources; they have invaded. Obtaining and sharing accurate information (e.g., taxonomic identification, pathological effects, and distributions) regarding these pathogens will be critical to help prevent their spread and to reduce the risk of other ANS.

Conclusion.—In the face of limited time and budgets, it is imperative that fisheries managers focus on strategic measures to achieve management objectives. Unfortunately, unforeseen events arise that challenge the best crafted plans. In the case of fish pathogens, challenges can influence salmonid stocks significantly. Although it may not be possible to be completely prepared, an understanding of potential threats should help improve salmonid health by (1) informing management decisions, (2) disseminating scientific research findings to adjacent resource managers who are likewise concerned with these salmonid pathogens, (3) hastening inter- and intrastate biosecurity measures, and (4) engaging in public outreach that targets anglers and other conservation-minded citizens.

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Parasites and the Health of Wild Trout: Should we be Concerned About *Salmincola edwardsii* Infecting Brook Trout?

Matthew G. Mitro¹, Joanna D. Griffin²

¹Wisconsin Department of Natural Resources, Office of Applied Science, Division of Fish, Wildlife and Parks, Science Operations Center, 2801 Progress Road, Madison, Wisconsin 53716 USA

²Wisconsin Department of Natural Resources, Fisheries Management Bureau, Division of Fish, Wildlife and Parks, 101 S. Webster Street, Madison, Wisconsin 53707 USA

Abstract—Changes in environmental conditions in coldwater streams have been implicated in parasitoses and associated declines in native wild trout populations. An epizootic of the ectoparasitic copepod *Salmincola edwardsii* in Ash Creek, Wisconsin in 2012-2014, for example, led to a 77-89% decline in age-0 Brook Trout *Salvelinus fontinalis* recruitment. Observations in Ash Creek and by anglers elsewhere raised concerns about *S. edwardsii* epizootics and their potential effect on Brook Trout in other Wisconsin streams. Here we provide a historical perspective on *S. edwardsii*, commonly known as gill lice, in Wisconsin; describe the current statewide distribution of *S. edwardsii* and its prevalence and intensity of infection in Brook Trout; and discuss individual- and population-level effects of *S. edwardsii* on Brook Trout and how models can be used to evaluate *S. edwardsii*-Brook Trout dynamics. In 2013-2016, Brook Trout were inspected in 214 streams across Wisconsin to determine the current statewide status of *S. edwardsii*. *Salmincola edwardsii* were present in 82% of streams and absent from 18%. Prevalence of infection ranged from 0.4% to 100% where the parasite was present. Intensity of infection of the most heavily-infected Brook Trout observed in a stream was light (1-5 *S. edwardsii*) in 36% of streams, moderate (6-14) in 35%, and heavy (≥ 15) in 29%. Intensity was heavy and prevalence exceeded 90% of Brook Trout in 3 of 214 streams, which suggests epizootics as observed in Ash Creek are uncommon. Models that account for susceptible versus infected states can be used to evaluate interactions among host, parasite, and environment and to evaluate factors conducive to epizootics, including host population density, parasite transmissibility, duration of infection, and development of virulence versus immunity. We conclude that concern for *S. edwardsii* infection of Brook Trout is warranted in heavily-infested populations, but that more research is needed on long-term *S. edwardsii*-Brook Trout dynamics to better understand the genesis and duration of epizootics.

Introduction

Anglers fishing Wisconsin trout streams have in recent years observed, with greater frequency, a parasite attached to the gills of Brook Trout *Salvelinus fontinalis*. Commonly called “gill lice,” the parasite is an ectoparasitic copepod properly identified as *Salmincola edwardsii* (Figure 1). This parasite has been documented in Wisconsin streams as early as the late 19th century (Fasten 1912). Successive generations of Wisconsin fisheries managers and researchers have observed the presence of the parasite in some Brook Trout populations but have generally not considered it a significant threat. However, in recent studies of Wisconsin Brook Trout populations, epizootics of *S. edwardsii* have been observed and population-level effects on Brook Trout have been documented (Mitro

et al. 2014; Mitro 2016). These epizootics may have been precipitated by changes in stream conditions including warmer temperatures and lower flows. For example, unseasonably warm temperatures in early spring 2012 and drought conditions through summer may have facilitated the *S. edwardsii* epizootic in Ash Creek, Wisconsin by providing conditions favorable to the *S. edwardsii* lifecycle (Mitro et al. 2014; Mitro 2016). *Salmincola edwardsii* prevalence in the Ash Creek Brook Trout population increased from 42% in April 2012 to 95% in October 2012, and the persistence of the parasite in the Brook Trout population through successive years was implicated in a 77%–89% decline in age-0 Brook Trout recruitment when compared to years prior to the epizootic (Mitro et al. 2014; Mitro 2016).



Figure 1. (a) Adult female *Salmincola edwardsii*; (b) heavily-infected Brook Trout; and (c) infected Brook Trout displaying hyperplasia and hypertrophy of secondary lamellae in the gills.

Should we be concerned about *Salmincola edwardsii* infecting Brook Trout? This is a question being asked more frequently as awareness of the parasite has increased. To answer this question, it may be helpful to consider this specific parasite-host system in a general sense. In his introduction to the study of parasitism, Combes (2001) compares parasite-host and predator-prey systems in relation to four types of wealth produced by organisms: their bodies, metabolism, work, and products of work. This wealth is coveted by other living organisms. A common example is the predator consuming the body of its prey, whereby energy is transferred from prey to predator. Parasites also exhibit a similar transfer of energy, here from host to parasite. What is different, however, is the duration of this exchange. Whereas the energy exchange between predator and prey is relatively instantaneous, the parasite-host interaction is prolonged and allows the parasite to exploit the host's wealth beyond the consumption of its body. The host becomes habitat for the parasite and their interaction may persist through the duration of the parasite life cycle, ending only when either host or parasite dies. If *S. edwardsii* causes premature death of its Brook Trout host, that may lead to a concern about the fishery. Short of host death, however, there may be sublethal effects of the parasite on the host (Vaughan and Coble 1975), which may also raise concern depending on whether the sublethal effects are expressed at the host population level.

An assessment of the potential threat *Salmincola edwardsii* may pose to Brook Trout populations necessitates information on the presence or distribution of the parasite among Brook Trout waters and on the prevalence and intensity of infection of Brook Trout where the parasite is present. Distributional records of *S. edwardsii* are limited, particularly at spatial scales useful to state or provincial management agencies. *Salmincola edwardsii* have been documented infecting Brook Trout in many locations across the Brook Trout's native range. The parasite is thought to be widespread in Wisconsin (e.g., Fasten (1912)), but their distribution and infection dynamics have not been quantified. *Salmincola edwardsii* has been documented as present in other states including Maine (Bonney 2007), Michigan (Muzzall 1984, 1986, 2007), North Carolina (Ruiz et al. 2017), Minnesota (R. Hoxmeier, personal communication), and Pennsylvania (J. Detar, personal communication). *Salmincola edwardsii* is

also present in waters of the Canadian provinces of Manitoba, Ontario, Quebec, New Brunswick, Nova Scotia, Labrador, and Newfoundland (Marcogliese and Cone, 1991; McDonald and Margolis, 1995). We are not aware of any efforts to characterize the distribution of *S. edwardsii* across streams at the state or provincial level. Studies on the prevalence or intensity of *S. edwardsii* on Brook Trout have also been limited in geographic scope, typically to a small number of streams (e.g., Black et al., 1982; Muzzall, 1984, 1986, 2007).

Here we begin to address the question about concern for *Salmincola edwardsii* infecting Brook Trout by quantifying the current statewide distribution of *S. edwardsii* and its prevalence and intensity of infection in Brook Trout in Wisconsin streams. We then discuss individual- and population-level effects of *S. edwardsii* on Brook Trout and how changing environmental conditions may influence such effects. Finally, we discuss how models can be used to evaluate parasite-host dynamics and address concerns about *S. edwardsii* infecting Brook Trout.

Methods

Brook Trout were inspected for *Salmincola edwardsii* in 214 streams across Wisconsin in 2013–2016. Streams were surveyed as part of Wisconsin’s statewide baseline monitoring of coldwater wadeable streams ($n = 201$) or for trout research projects ($n = 13$).

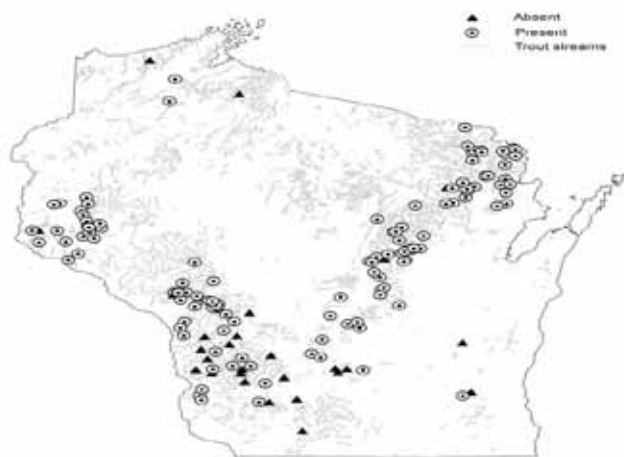


Figure 2. Presence (circle-dot) and absence (shaded triangle) of *Salmincola edwardsii* in surveyed Wisconsin streams where Brook Trout were observed ($n = 153$). Fifteen or more Brook Trout were inspected in streams in which *S. edwardsii* were documented as absent.

Stream survey sites were about 35 times mean stream width or longer in length, but not less than 100 m, and typically included more than three riffle-pool sequences. Survey site lengths ranged from 100 to 1,980 m (lower quartile = 100 m, median = 140 m, and upper quartile = 252 m). The number of survey sites per stream ranged from 1 to 13 ($n = 356$). We combined all sites within a stream for analyses, and total length of combined survey sites by stream ranged from 100 to 3,309 m (lower quartile = 109 m, median = 200 m, and upper quartile = 400 m).

Brook Trout and Brown Trout *Salmo trutta* were collected at each site with either a single-pulsed DC backpack electrofisher or a tow barge electrofisher with two or three hand-held anodes. All observed trout were collected. We asked biologists who collected data on *S. edwardsii* infection of Brook Trout as part of baseline stream monitoring surveys to inspect at least 60 age 1 and older Brook Trout per site or all age 1 and older Brook Trout if fewer than 60 were captured. We inspected all age 1 and older Brook Trout captured as part of trout research projects. Biologists were also asked to describe the intensity of infection for the Brook Trout they observed with the most *S. edwardsii* at each survey site: light (1–5 *S. edwardsii*), moderate (6–14), or heavy (15 or more). We used data from the most recent year for sites in streams that were surveyed in multiple years.

We identified *Salmincola edwardsii* as present in a stream if at least one Brook Trout was observed infected. We considered *S. edwardsii* absent if no infected Brook Trout were observed and at least 15 Brook Trout were inspected among all sites in a stream. Prevalence of infection was defined as the percentage of inspected Brook Trout infected with *S. edwardsii* and was calculated when at least 15 Brook Trout were inspected among all sites in a stream. Confidence intervals for prevalence estimates were calculated using the Wilson score interval (Zelmer 2013). Intensity of infection for a stream was the highest level of infection intensity observed in a Brook Trout among all sites in a stream in which *S. edwardsii* was present.

Results

We observed *Salmincola edwardsii* infecting Brook Trout and documented them as present in 125 streams across Wisconsin (Figure 2). There were 28 streams in which 15 or more Brook Trout were

captured and inspected and *S. edwardsii* were not observed and therefore documented as absent. There were 61 streams in which Brook Trout were present but fewer than 15 were inspected and no *S. edwardsii* were observed; we did not draw any conclusion on the presence or absence of *S. edwardsii* in these streams. *Salmincola edwardsii* were present in 82% of streams and absent from 18% for the 153 streams in which the presence or absence of *S. edwardsii* was determined.

Prevalence of infection ranged from 0.4 to 100% (n=86) in streams where *S. edwardsii* were present and 15 or more Brook Trout were captured and inspected (Figure 3). Less than one-third of Brook Trout were infected in 23% of streams (n=20), one to two-thirds were infected in 49% of streams (n=42), and over two thirds were infected in 28% of streams (n=24) (Figure 4). Prevalence of infection exceeded 90% of Brook Trout in four streams.

We determined the intensity of infection for the most heavily-infected Brook Trout observed in a stream for 122 streams in which *S. edwardsii* were observed and counted. Intensity of infection was light in 44 streams (36%), moderate in 43 streams (35%), and heavy in 35 streams (29%) (Figure 5). Intensity of infection was heavy and prevalence of infection exceeded 90% of Brook Trout in three streams (Figure 6).

Discussion

We found *Salmincola edwardsii* to be widespread across trout streams in Wisconsin, with varying levels

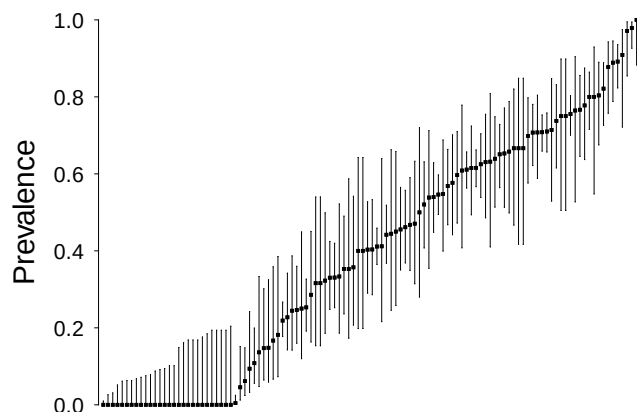


Figure 3. Prevalence of *Salmincola edwardsii* (95% confidence intervals) in surveyed Wisconsin streams in which 15 or more Brook Trout were inspected (n = 114).

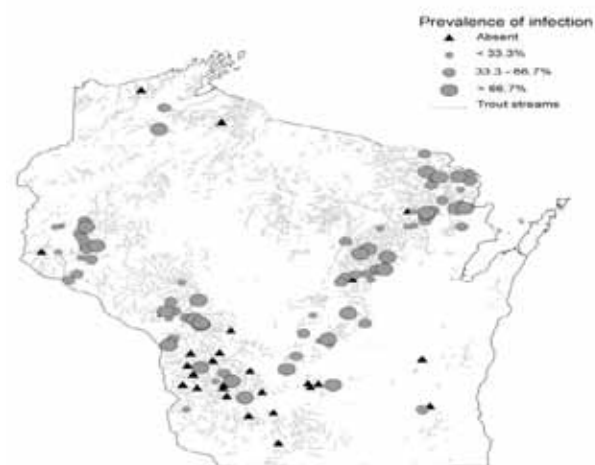


Figure 4. Prevalence of *Salmincola edwardsii* in surveyed Wisconsin streams in which 15 or more Brook Trout were inspected (n = 114).



Figure 5. Intensity of *Salmincola edwardsii* infection in surveyed Wisconsin streams in which 15 or more Brook Trout were inspected (n = 122).

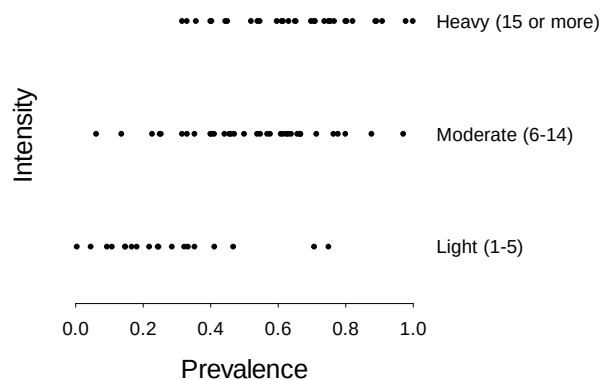


Figure 6. Prevalence versus intensity of infection of *Salmincola edwardsii* in surveyed Wisconsin streams where infected Brook Trout were observed (n = 86).

of prevalence and intensity of infection showing no distinct geographic patterns. Some populations of Brook Trout appeared to be free of *S. edwardsii* whereas nearby populations were heavily infested. Such populations, though close in proximity to one another, may lack connectivity. One of the largest uninfected Brook Trout populations, in the South Fork of the Kinnickinnic River in western Wisconsin, has a waterfall that acts as a barrier to upstream movement from other connected waters. Some populations with uninfected Brook Trout occur in streams in which Brook Trout had previously been extirpated and reintroduced by stocking. These streams often have barriers to movement in the form of poor physical or thermal habitat.

Salmincola edwardsii, like other ectoparasitic copepods, tend to aggregate on their hosts (Poulin et al. 1991; Poulin 2007). A small number of Brook Trout in an infested population may be infected by many *S. edwardsii* whereas most Brook Trout have few or none. We collected data on intensity of infection by noting, for each population, the host with the largest aggregate of *S. edwardsii* and categorically assigning a level of intensity of infection. Brook Trout with 15 or more *S. edwardsii* were considered heavily infected. Although this approach did not quantify the numerical range of *S. edwardsii* that can occur on a heavily-infected host, it was an approach amenable to working with live fish in a field setting. For heavily-infected brook trout with 15 or more *S. edwardsii*, accurate counts could only be obtained in a laboratory setting with sacrificed fish. Mitro (2016) observed aggregates as high as 97 *S. edwardsii* infecting an adult Brook Trout in Ash Creek, Wisconsin. Aggregation probably occurs through a combination of heterogeneity in exposure, susceptibility to infection by *S. edwardsii*, and chance (Poulin 2011). Of particular concern will be populations in which large numbers of *S. edwardsii* aggregate on large numbers of Brook Trout. This pattern of infection has persisted at long-term trend sites in Ash Creek, where in April 2017 the infection prevalence was 91% (70 of 77 Brook Trout infected) and 42% of Brook Trout had 15 or more *S. edwardsii* (32 of 77 Brook Trout).

Heavy intensity of infection tended to occur in populations with higher levels of prevalence. Conversely, light intensity of infection tended to occur in populations with lower levels of prevalence. There were no heavily infected Brook Trout in populations

with prevalence of infection less than 31%. And the intensity of infection was moderate or heavy for the Brook Trout observed with the most *S. edwardsii* in all populations with prevalence levels greater than 75%. There was considerable overlap in the level of infection intensity observed at different levels of prevalence.

Some of the physical effects of *Salmincola edwardsii* parasitism on Brook Trout are well documented (Vaughan and Coble 1975). *Salmincola edwardsii* typically attach to gill filaments, causing physical trauma affecting the respiration process and potentially leading to mortality and declines in host populations (Piasecki et al. 2004; Vigil et al. 2016). Immune response by Brook Trout to infection includes epithelial hyperplasia and hypertrophy of gill tissue, which is a rounding, shortening, and fusion of secondary lamellae (Figure 1). This damage to the gills may affect the efficient uptake of dissolved oxygen and the release of metabolites such as carbon dioxide and ammonia (Conley 1994). Heavily infected Brook Trout may not be able to obtain sufficient dissolved oxygen when they are exercised, such as when caught by angling or when spawning. Respiration may be particularly difficult for infected fish during times of high water temperatures and low dissolved oxygen levels. Infection may also affect fish behavior, leading infected fish to become more susceptible to further infection (Poulin et al. 1991).

There is little information available on Brook Trout mortality attributable to *S. edwardsii*. Mitro et al. (2014) and Mitro (2016) showed a decline in Brook Trout recruitment in a heavily infested population. *Salmincola edwardsii* were a probable factor in the recruitment decline, but the mechanism linking *S. edwardsii* to the decline has not been defined (e.g., direct mortality of age-0 Brook Trout versus sublethal effects that increase their susceptibility to other forms of mortality).

A study of the infection of Arctic Charr *Salvelinus alpinus* by the sea louse *Lepeophtheirus salmonis* may provide some insight on the potential effects of *Salmincola edwardsii* infection of Brook Trout. Tveiten et al. (2010) infected groups of mature (age 5 and older) Arctic Charr with *L. salmonis* for 34 d during early stages of gonad development to test for effects on changes in cortisol, sex steroids, growth, and reproductive investment. Infection levels were classified as high (0.15 mean number of

L. salmonis/g of fish mass), medium (0.07 mean number of *L. salmonis* g of fish mass), or none (control). Intensity of infection positively influenced cortisol concentrations and negatively influenced plasma sex-steroid concentrations. Growth and condition of the most heavily infected Arctic Charr were negatively influenced by infection, and reproductive development was postponed. Mortality was 40% in the high infection group versus 6% in the medium infection and none groups. There were lower numbers of maturing females in the high and medium infected groups with egg production about 50% and 30%, respectively, of that in the control group. However, egg size, embryonic survival, and fry mass did not differ among groups. These study results suggested that population-level effects of *L. salmonis* infection of Arctic Charr in the form of sublethal effects on mature females led to reduced growth and reduced reproductive output. We are not aware of any similar studies concerning *S. edwardsii* infection of Brook Trout. However, observed levels of infection in Ash Creek Brook Trout in 2012 were greater than the high infection group in the Tveiten et al. (2010) experimental study: age 1 and older Ash Creek Brook trout had a mean of 0.22 *S. edwardsii* per gram fish mass (95% CI = 0.16-0.27; maximum = 0.99; Mitro (2016)). If the Brook Trout response to *S. edwardsii* infection is similar to the Arctic Charr response to *L. salmonis* infection, then population-level effects may be realized via reductions in reproductive output.

Considering the effects of infection at the population level may be required to answer the question about whether or not we should be concerned about *Salmincola edwardsii* infection of Brook Trout. If *S. edwardsii* affects mortality or fecundity of Brook Trout, then there will be a population-level effect commensurate with the magnitude of change in host mortality or fecundity. The population decline described in Mitro et al. (2014) and Mitro (2016) occurred in a Brook Trout population with a high prevalence of infection (> 90%) and high intensity of infection (many fish infected with ≥ 20 *S. edwardsii*). The presence of a sympatric population of Brown Trout, which were not infected by macroparasites, was an additional factor threatening the dynamics of the infected Brook Trout population. In sympatric populations of native Brook Trout and introduced Brown Trout, *S. edwardsii* epizootics may hasten the displacement of Brook Trout by Brown Trout (Mitro

et al. 2014; Mitro 2016). The population decline in Ash Creek is very concerning, but our study on *S. edwardsii* distribution in streams across Wisconsin showed that the high prevalence and high intensity of infection observed in Ash Creek were not common elsewhere. Long term studies are needed to better understand *S. edwardsii*-Brook Trout dynamics and the onset and duration of epizootics. Models may be useful in exploring such dynamics and thresholds for epizootics.

Parasitosis is a dynamic phenomenon with fluctuations in parasite-host populations depending on the host, parasite, and environment. Models can describe parasite-host dynamics by parameterizing population processes of parasite and host and environmental factors affecting parasitosis (Anderson and May 1978; Reno 1998). *S-I-R* models, for example, describe the host population in terms of individuals susceptible to infection (*S*), infected (*I*), or removed from the population by mortality (*R*). Host and parasite numbers change in response to the processes of birth, death, immigration, and emigration, and infection dynamics may change in response to environmental conditions such as temperature. Genetic changes in *S. edwardsii* virulence versus Brook Trout immunity may also change the ratios of susceptible and infected hosts in the population. The rate of parasite transmission and the basic reproductive rate of the parasite determine the status of parasitosis. Basic reproductive rate R_0 of *S. edwardsii* is a product of transmission efficiency β , population density, and the duration of infection. If β and R_0 are low, *S. edwardsii* does not infect enough hosts to persist and may become eradicated from the host population. As β and R_0 increase, *S. edwardsii* may become enzootic. And if β and R_0 increase high enough, an epizootic will occur and *S. edwardsii* may significantly regulate population size depending on the severity of the epizootic.

Matrix population models can also be used to quantify potential effects of *S. edwardsii* on Brook Trout population growth rate and to identify the sensitivity of population growth rate to changes in population vital rates attributable to *S. edwardsii* infection. Models have suggested that Brook Trout are capable of rapid population growth but are sensitive to changes in survival at age-0 to age-1 (Heppell et al. 2000; Peterson et al. 2008). This model observation is consistent with field observations of the infection of age-0 Brook Trout in Ash Creek and the decline

in recruitment attributable to losses in that age class (Mitro et al. 2014; Mitro 2016). However, more data will be needed on *S. edwardsii* and Brook Trout population vital rates and the effect of environment on infection dynamics to better evaluate effects of *S. edwardsii* on Brook Trout population growth rates.

Our data on *Salmincola edwardsii* distribution, prevalence, and intensity of infection across Wisconsin streams suggest that the concern about Brook Trout loss attributable to *S. edwardsii* as observed in Ash Creek may only be warranted in a small percentage of streams with high prevalence and intensity of infection. That said, more data are needed on long-term *S. edwardsii*-Brook Trout dynamics to better understand the genesis and duration of epizootics. Data on environmental factors such as stream temperature, flow, and sympatric populations of other trout species that may compete with Brook Trout are also critical to evaluating how *S. edwardsii* and Brook Trout change in abundance over time and to help guide management actions for preventing or addressing *S. edwardsii* epizootics. Research is also needed to test whether there is a genetic basis for immunity in populations not infected with *S. edwardsii*.

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Insights into the Evolution of *S. trutta* Major Histocompatibility Complex Across Populations under Parasite Infection

Toby Landeryou

Middlesex University, London, UK

Major Histocompatibility Complex Structure and Function

The primary role of the major histocompatibility complex (MHC) is to recognise foreign proteins, present them to specialist immune cells and thus initiate an immune response. Recognised foreign proteins are broken down into their peptides and interact with specific MHC molecules (Kurtz et al. 2004; Aguilar & Garza 2006), which are then transported to circulating T-cell population, initiating a complex cascade of immune responses (Perrigoue et al. 2010). The MHC molecule comprises of an immunoglobulin stalk, which is anchored to the cell surface. To recognise the foreign pathogen the structure also includes a “basket” receptor called the “antigen binding site” (ABS, also called the ‘peptide-binding groove’, ‘peptide binding site’ or ‘antigen recognition site’) (Cuesta et al. 2006; Forsberg et al. 2007; Spurgin et al. 2010; Kamiya et al. 2014). The ABS is responsible for antigen recognition, the interaction between the ABS, antigenic peptide and the T-cell receptor is required to complete a significant immune response.

The MHC consists of a group of closely linked genes that are some of the most important genetic components of the vertebrate immune system. Genes of MHC class II are predominantly expressed on the surface of all nucleated somatic cells due to the importance of monitoring the extracellular environment for pathogens (Dionne et al. 2008). Within class II genes, most research is focused on a series of genes that encode exon 2 of the MHC, because these genes code for functionally important antigen binding sites (ABS) (Froeschke and Sommer 2004; Blais et al. 2007; Lenz et al. 2009; Schewensow et al. 2010; Zhang et al. 2015).

Parasite Mediated Selection

A large body of empirical data indicate that positive selection acts on MHC loci to maintain MHC variation within populations (Radwan et al. 2014; Bracamonte et al. 2015; Perchoukova et al. 2015; Schuster et al. 2016).

When selection occurs in a population, the genes under positive selection will show an increased rate of nucleotide substitution relative to neutral expectation (Bernatchez & Landry 2003). These effects can be detected within a population using several approaches. The ratio of non-synonymous (amino-acid altering) to synonymous (silent) nucleotide substitutions per non-synonymous and synonymous site (dN:dS) can be used to test for positive selection (Hughes and Nei 1988). Synonymous substitutions do not affect the amino-acid composition of the final translated gene sequence. Substitutions that affect the final compositions of amino acids are non-synonymous substitutions; these are more likely to be under selection (Spearman and Wilke 2015). When considering selection occurring within the ABS of MHC genes, if diversity was favoured within the region advantageous non-synonymous mutations will be retained resulting in dN:dS value being above 1 (Sabati et al. 2010). Research involving tests of deviation of dN:dS from neutral is becoming increasingly common and now examine which amino-acid sites present the strongest signals of selection (Suzuki 2004; Massingham and Goldman 2005). These approaches have proved useful in studies of the MHC, as the effects of selection on specific known antigen binding regions can be ascertained.

MHC of Fish

Several mechanisms of balancing selection ensure MHC genes are the most polymorphic genes within vertebrates (Piertney and Oliver 2006). As with other vertebrates, the allelic diversity observed within natural fish populations gives them the ability to defend against a range of parasites within the spectrum both in terms of specific parasitic alleles and the diversity of infective parasite species (Bracamonte et al. 2015). Communities dominated by a generalist parasite may represent fluctuating genotypic diversity compared to a population where parasite specific MHC haplotypes enable efficient sorting of resistance within the population (Monzon-Arguello et al. 2014); this is why a less specific, broader individual

MHC diversity are expected to be more important in organisms that are likely to live within a multi-parasite threat milieu. This only gives fish host's resistance to a limited range of parasite infections, promoting host-parasite co-evolution because it provides potential for new parasite antigens to evade detection. In cichlids and *Gasterosteus aculeatus* the number of MHC loci used differs between individual haplotypes (Málaga-Trillo et al. 1998). Other studies describe numbers of MHC alleles differing due to parasite community richness both within a population (Wegner et al. 2003; Göüy De Bellocq et al. 2008) and within individuals (Tobler et al. 2014; Bracamonte et al. 2015).

Salmonid genetic variability of the MHC locus has been linked to kin discrimination (Olsen et al. 1998), mate choice (Forsberg et al. 2007), embryo viability (Jacob et al. 2010), pathogen infection (Consuegra & Garcia de Leaniz 2008; Lamaze et al. 2014) and can also play an important role in the viability of invasive species or nonnatives strain introduction (O'Farrell et al. 2013; Monzon Arguello et al. 2014). The genetic variability of the salmonid MHC complex clearly has numerous ways to ensure diversity. The increased diversity of the locus has beneficial effects on population fitness, with empirical evidence suggesting that parasite-driven selection being a key component. With the exact selection mechanism under debate (Reusch et al. 2001; Wegner et al. 2003; Stutz & Bolnick, 2016), the high variability and adaptive significance of MHC genes make them an ideal candidate to study selective pressures pathogens exert on host immune systems.

Hill loch systems contain fractured and varied populations of salmonids. Within the Gairloch sampling sites, populations of post-Pleistocene Brown Trout *Salmo trutta* are spread throughout freshwater bodies with varying degrees of isolation. These populations have adapted to these water bodies with the adaptation to possible parasite infection being a major factor affecting cohort fitness. This research goal is to quantify the role that selective pressure associated with parasite infection plays on salmonid population immunogenetics. This will inform development of sustainable, captive breeding strategies within aquaculture and conservation. Setting out and completing the following objectives will achieve this goal:

1. Amplification of MHC II β associated gene across the Wester Ross Trout Fishery trusts trout archive,

2. Phylogenetic construction to ascertain possible geographic structuring of the gene within the hill loch system,
3. Diversity and selection analysis of the gene throughout the population, and
4. Post-translational analysis on the gene within differential parasite infection environments to ascertain species-specific selection acting on MHC

Methodology

Sample Collection

Whole fish samples were collected from the years of 2011-2014, donated by anglers in conjunction with the Wester-Ross Fisheries Trust. A total of 185 whole Brown Trout were used in amplification of *Satr-DAB*. The use of gene *Satr-DAB* in analysis was due to the gene encoding the antigen-binding site within MHC II in Brown Trout. Tissue samples were taken internally to ensure no cross contamination, and were stored within individual Eppendorfs within 70% ethanol before DNA extraction.

PCR Protocol

Total genomic DNA was extracted using protocol of Qiagen™ Blood and Tissue extraction kit. A 252 bp fragment of exon 2 of the MHC II locus, which exists as a single copy in salmonids including Brown Trout (Jacob et al., 2010), was amplified and sequenced. The MHC II locus is called *Satr-DAB* in *S. trutta*. The primers used were CL007 5'-GAT CTG TAT TAT GTT TTC CTT CCA G-3' and AL1002 5'-CAC CTG TCT TGT CCA GTA TG-3' (Olsen et al., 1998). PCR reactions were performed for each gene fragment using 12.5 μ l hot start taq polymerase; Thermo-StartRPCR master mix (0.625 Units of Taq DNA polymerase, 1X reaction buffer, 0.2 mM of each dNTP and 1.5 mM MgCl₂), 5 ng/ μ l of DNA and 3.5 ng/ μ l of each primer. The template PCR program used for all isolates was: 95° C for 15 min; 40 cycles at 95° C, 1 min; 55-58° C; 72° C, 1 min; 72° C for 7 min. Annealing temperatures were optimized on the primer set as 52° C.

Genetic Diversity and Selection – MHC II β

Sequences were edited and alignment using Geneious v6.5. Diversity data was acquired through the use of DNAsp software package (Librado and Rozas, 2009), developed for comprehensive analysis

of DNA polymorphism data. The analysis of codon-by-codon selection was performed using Hyphy population genetics program (Pond and Frost, 2005) this was performed at protein level using codon specific dN/dS. Geographic structuring was inferred using via Maximum likelihood (ML) method using Bayesian inference criterion (BIC) in Beast bayesian analysis software. The software allows for accurate phylogenetic tree construction using a MCMC (Markov chain posterior probability analysis) for testing evolutionary hypothesis without conditioning to a singletree topology. These were general time reversible models including estimates of site-rate heterogeneity (GTR+G), models were inferred using JModelTest9 (Diarriba, 2002).

Translation and Amino Acid Modelling

The reference *Satr-DAB* LFD 13 allele was submitted to SWISS-MODEL. The model used was murine MHC I 2bvoA with which *Satr-DAB* LFD 13 showed 59% similarity. The returned Protein Data Bank (PDB) files were loaded into the supplied SPDV DEEPVIEW program for three-dimensional visualisation, graphical manipulations, and the plotting of codons under different selective pressures. SPDV DEEPVIEW was used to output files for the rendering software POV-RAY, which produces very high quality graphics of the protein. Consensus amino profile divided into two separate groups for means of selection profiling. Elevated Lochs were denoted consensus sequences from LAM; Loch Arigh Mi Criadh, LAP; Loch Arigh a'Phaill, LDA; Loch Doire na-h Arigh and LFM; Loch Feithe Mugaig. The other group used for comparison of dN/dS data was denoted 'Western Lochs', these consisted of LNB; Lochan nam Breac, LCA; Loch Coire na-h Arigh, LGD; Loch Gharbe Doire and LFE; Loch Feur.

Results

Phylogenetic Analysis and Diversity

Phylogenetic tree construction revealed very little geographic differentiation among individual Loch populations (Figure 1). The only distinct population lineage was within Loch a'Mhadaidh. Clade topology was not related to parasite specific species infection. The gene showed diversity with 168 haplotypes occurring within a sample size of 185. Diversity analysis across the system indicates

the highest amount of diversity occurring within Loch Fada (LFD) (K; 18.133) along with the largest amount of segregating sites (S;46). The Lochs experiencing the lowest population diversity was Loch na h-oidche(LNO) (K; 5.21795) which also had the lowest amount of segregating site (S; 17). In regards to dN/dS the highest values were within Loch Gharbe a'Doire (LGD) (dN/dS; 3.9044), Lochan nam Breac (LNB) (dN/dS; 4.1647) and Loch Feur (LFE 4.054). All populations indicated positive selection acting on *Satr-DA*, with every population dN/dS>1.

Amino Acid Analysis

Translated protein sequencing was performed on 40 samples from western lochs and 40 samples from elevated lochs divided by parasite species infection acquired from parasite screen (Table 1). Consensus protein sequences were created for the two sequence sets (western and elevated lochs) and compared according to dN/dS value by individual codon-by-codon basis. Within western loch populations a total of 13 out of a possible 43 amino acids were under positive selection (dN>1), whilst elevated loch population consensus sequences showed 8 sites under positive selection out of a possible 43 (Figure 2). Within 3D construction of both consensus protein sequences of western and elevated protein structures show proteins under positive selection within hypothesised peptide binding site of the protein structure (Figure 3). Proteins, which express the highest value of dN/dS within western loch consensus, include Lysine (codon 70, dN/dS; 8.7) and Arginine (codon; 172, dN/dS; 8.4569). Within elevated loch consensus protein sequence Lysine (codon; 64, dN/dS; 6.54) and Arginine (codon;178, dN/dS; 9.289) were also the two codons with the highest value for dN/dS.

Discussion

Genetic diversity of *Satr-DAB* showed clear inter-population differentiation. One constant throughout the Gairloch system was the strong signal of positive selection acting on the MHC *IIβ*, with all lochs showing dN/dS>1 on *Satr-DAB* gene. The dN/dS values were corrected for using p value, with normalized dN/dS averaging 3.3. It is reasonable to assume immune system genes would be under strong selective pressure. Positive selection indicated through elevated dN/dS values have been demonstrated in

Table 1. Diversity data associated with *Satr*-DAB gene across Scottish hill loch populations - Number of segregating sites, **Pi** – nucleotide diversity, **K** – average number of nucleotide diversity per site, **h** – number of haplotypes, **hd**– diversity between haplotypes observed.

Site	length	S	hap	Hd	K	Pi	samples	dN	dS	dN/dS
LFD	243	46	9	0.97778	18.133	0.12009	10	120.65	38.35	3.146
LDB	243	28	7	0.96429	9.96429	0.06599	8	125.83	39.17	3.2124
LNU	243	33	5	1	15.7	0.10387	5	124.43	37.57	3.3119
TWA	243	23	7	1	8.52381	0.05645	7	117.21	38.79	3.0216
LMD	243	28	9	1	9.55556	0.06328	9	117.13	35.87	3.2654
AAG	243	24	9	0.96364	7.8	0.05166	11	116.14	33.86	3.43
LCA	243	30	9	0.97165	9.67949	0.0641	13	111.45	32.55	3.4239
FES	243	23	6	1	10.6	0.0702	6	116.81	36.19	3.2276
LAP	243	29	17	0.98421	6.55789	0.04343	20	113	34	3.3235
LGD	243	30	11	0.95604	10.17582	0.06739	14	156.02	39.96	3.9044
LFE	243	36	12	0.96884	12.01606	0.07266	9	159.85	39.07	4.054
LNB	243	42	10	0.95455	14.07576	0.09322	12	158.22	38.56	4.1647
LFM	243	18	6	0.8	6.94545	0.046	11	118.2	37.8	3.1269
LNO	243	17	9	0.87179	5.21795	0.03456	13	118.79	37.21	3.1924

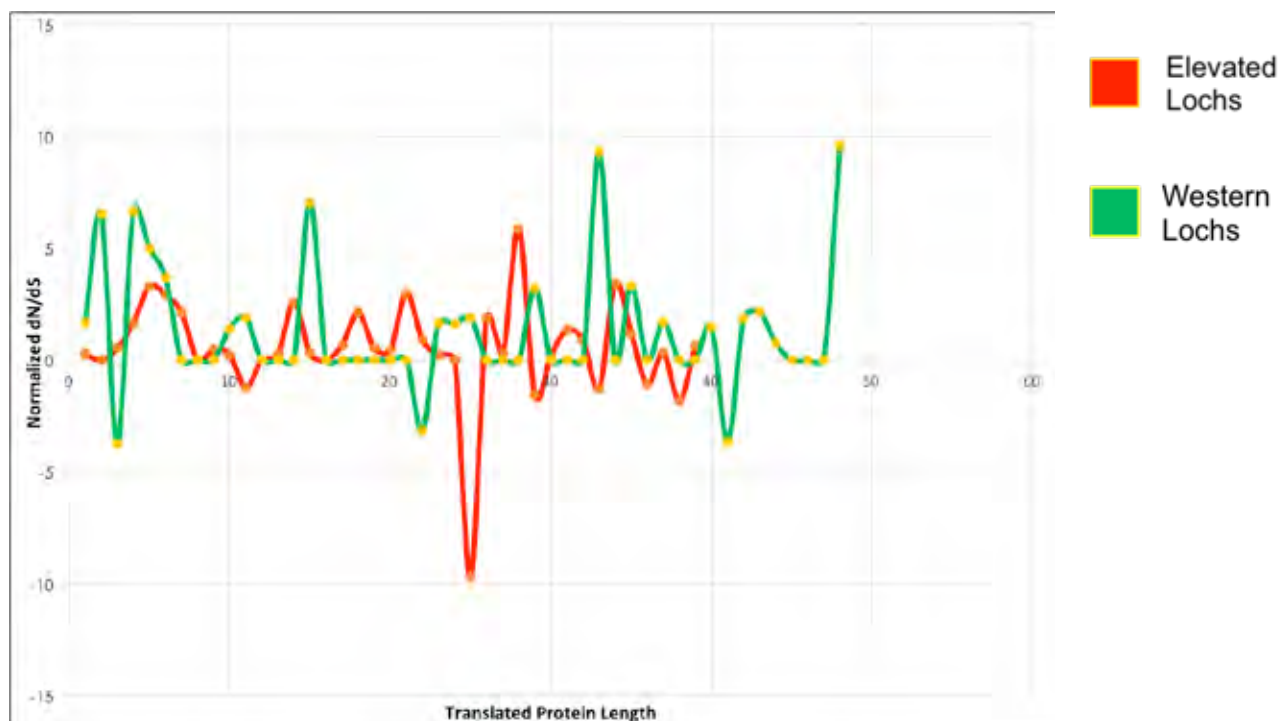


Fig 2. dN/dS value across length of consensus amino acid sequence for Western Lochs and Elevated Lochs, each point along graph denotes single amino which is part of consensus amino sequence for each location.

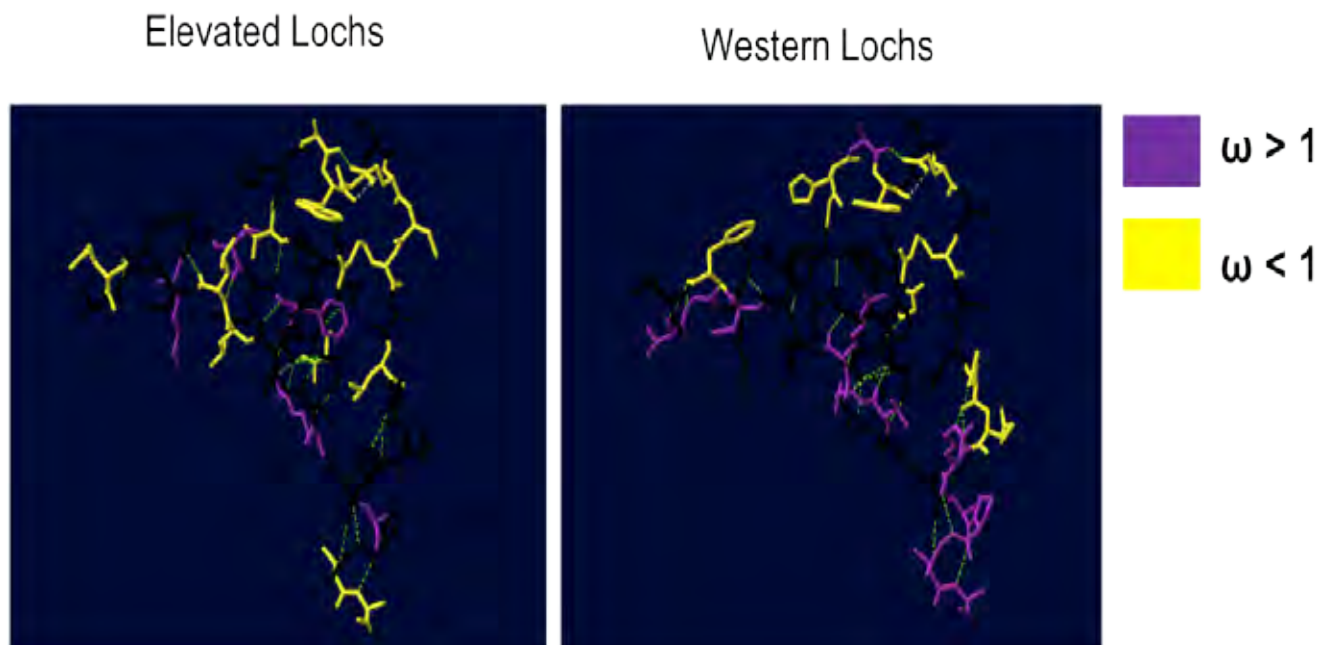


Fig 3. 3D protein analysis of consensus sequence within Western and Elevated locale populations. Colour denotes proteins within 3D structure that are under positive selection **purple**; $\omega > 1$ ($dN/dS > 1$), **yellow**; $\omega < 1$ ($dN/dS < 1$).

numerous other taxa (Radwan et al. 2014; Zueva et al. 2014; Wezner et al. 2016).

Phylogenetic reconstruction revealed little to no geographic structuring of different populations across the region suggesting that although selective pressure is strong, it has not lead to significant differentiation in terms of topology. The only population that did show isolation in terms of phylogenetic construction was Loch a'Mhadaidh, the farthest west loch sampled isolated from potential anadromous migration via an impassable waterfall between the loch and marine environment. The population with the greatest diversity was Loch Fada (K; 18.133), one of the largest by area (3.46 km²) implying presence of a large trout population with increased diversity resulting from a larger gene pool compared to other, smaller lochs. Population size is not the only contributing factor towards genetic diversity of *Satr*-DAB with the western lochs; Lochan nam Breac (0.05km²) and Loch Feur (0.04km²) compared with larger (e.g. Loch Feithe Mugaig 0.16km²) elevated lochs demonstrated greater genetic diversity in respect to *Satr*-DAB.

The increased diversity of the western lochs could be explained by the elevated value of dN/dS with the MHC parasite derived selective pressure due to increased parasite load and diversity of infective species in western lochs.

Tests at both the genomic and protein level indicated the presence of strong signals of positive selection. These signals were more pronounced within populations that were infected by multiple parasite species with a larger proportion of codons under positive selection within western loch populations. The pattern of increased selection within the ABS can be interpreted as a result of pathogen-driven selection leading to high level of intra-population diversity associated within increased diversity of pathogenic insult (Sommer 2006). The individual codon analysis supported a number of codon positions under strong levels of selection compared to other codons. In contrast to Elevated Lochs the Western Lochs displayed differential codons under high levels of selection. Analysis did not indicate that any of the codons under positive selection within western Lochs shared with the Elevated Lochs.

The wild Brown Trout populations within the study are highly fragmented and found within isolated locations with wholly different environmental characteristics in respect to hydrology, fauna and potential pathogenic infection. One of the factors, which Brown Trout in the Gairloch system need to adapt to, is the clear differential parasitism exhibited within the hill loch populations. The predominant species infecting Brown Trout within the Gairloch database is the eye fluke *Diplostomum baeri*; however, the effect of cestode infection within the Western Lochs provides a clear potential impact upon trout inhabiting the lochs in terms of fitness (Table 1) (Urquart et al. 2010). Within smaller isolated populations of wild Brown Trout it might be expected that populations may be more influenced via genetic drift rather than balancing selection due to the size of population (Jones 2003); however, in regards to Gairloch hill loch populations parasite-driven selection clearly plays a role, with differing selection acting upon immune related genes in response to parasitic fauna. Clearly, part of the adaptation is ensuring that parasitic fauna does not have a negative affect on the fitness of the population. Differential pathogen infection not only has selective pressures toward *Satr-DAB*, the gene encoding the MHC II but also affects the final amino structure of the Antigen binding site, which could infer conformational change, creating differential binding capabilities with pathogen peptides increasing or decreasing the successful host defence of parasite infection.

This research presents data that helps to explain the complex evolutionary host-parasite relationship of salmonids inhabiting a hill loch system. Understanding the way salmonids adapt to minimise potential fitness affects posed via parasite infection should be considered with potential stocking procedures of nonnative fish or ensuring aquacultural processes are successful

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Air Exposure and Fight Times for Anadromous Fisheries in Idaho

Luciano V. Chiaramonte^{1*}, Don W. Whitney², Joshua L. McCormick¹, and Kevin A. Meyer¹

¹Idaho Department of Fish and Game, 1414 E Locust Ln, Nampa ID 83686

²Idaho Department of Fish and Game, 3316 16th St, Lewiston, ID 83501

*Corresponding author: luciano.chiaramonte@idfg.idaho.gov

Abstract—Increasing concern exists about the effects of air exposure and fight times on fish being caught and released. Such effects are usually tested in laboratory or hatchery settings, with little knowledge of actual angler behavior. We measured air exposure and fight times by anglers catching and releasing fish in popular salmon and steelhead fisheries in Idaho, and recorded other relevant factors such as fishing gear (fly or non-fly), occurrence of anglers photographing their catch, and landing method (net or hand). Overall air exposure time for fish caught and released averaged 28.7 s (3.2). Air exposure time did not differ with gear type but was 1.53 times (~15 s) longer if the angler took a photo of their catch. Fight time averaged 131 s (10.6) and differed with gear type, with fly anglers taking 1.57 times (76 s) longer to land fish than non-fly anglers. Deep hooking rate was 0% for fly ($n = 40$) and bait/jig terminal tackle ($n = 49$), and 1% for lures ($n = 99$). In the context of previous studies that have measured mortality of salmonids, the effects of these fight and air exposure times and deep hooking rates are likely negligible, particularly from a population-level perspective.

Introduction

The potential effects of catch-and-release angling on fish mortality has been a subject of extensive research for decades (see reviews by Muoneke and Childress [1994] and Bartholomew and Bohnsack [2005]) and concerns over sublethal physiological effects and general fish welfare are growing research areas (Davie and Kopf 2006; Huntingford et al. 2006; Arlinghaus et al. 2007; Cooke and Sneddon 2007). Aspects of catch-and-release angling that have been shown to affect post-release performance and survival include (among others) terminal tackle type (bait, lure, fly), fish handling (fight time and air exposure), and environmental conditions (water temperature and capture depth).

Of these factors, the effect of air exposure and fight times on post-release stress response and mortality has received perhaps the most attention in recent fisheries literature (reviewed in Cook et al. 2015). Studies conducted in laboratory or hatchery settings have generally shown negative effects of air exposure and fight times on fish (Ferguson and Tufts 1992; Schisler and Bergersen 1996; Schreer et al. 2005; Donaldson et al. 2014; Gale et al. 2014). However, taken collectively, this and other research suggests that, in most instances and for most species, unless released fish are air exposed for a minute or longer, long-term impacts are rarely life-threatening. Nevertheless, some states have been pressured by

special interest groups to enact regulations prohibiting air exposure of caught-and-released fish for some species. For example, in Washington, it is now unlawful to remove salmon, steelhead *Oncorhynchus mykiss*, or Bull Trout *Salvelinus confluentus* from the water if the angler subsequently releases the fish. With regard to fight time, exhaustive exercise has been implicated as having negative consequences for caught-and-released fish (Ferguson and Tufts 1992; Schreer et al. 2001), but as with air exposure, such impacts typically do not materialize unless fight times are exorbitantly long.

Considering the breadth of research on the effects of various levels of air exposure and fight time on caught-and-released fish, surprisingly little information exists about actual angler behavior in catch-and-release fisheries, and whether fight and air exposure times are problematic. In Idaho, covertly measured angler observations at several lake and river fisheries revealed that trout anglers held fish out of the water on average for 26 s before releasing them, with only 4% of anglers holding fish out of water for >60 s; fight time averaged only 53 s (Lamansky and Meyer 2016). While such fight and air exposure times likely have a negligible impact on hooking mortality for caught-and-released trout, the extent to which these findings apply to other fisheries is unknown. Our primary study objective was to evaluate fight and air exposure times in a popular Idaho catch-and-release fishery for salmon and steelhead.

Hooking mortality associated with catch-and-release fishing for anadromous species in the Pacific Northwest is attributed, in part, to terminal gear type and anatomical hooking location (Hooton 1987; Bendock and Alexandersdottir 1993; Lindsay et al. 2004; Nelson et al. 2005; Cowen et al. 2007), though other factors such as hook size and type, species, fish size, and capture conditions are important. Bait fishing generally results in higher rates of deep hooking, which more often injures vital internal organs or the gills, and frequently leads to higher rates of catch-and-release mortality (Muoneke and Childress 1994; Bartholomew and Bohnsack 2005). Because deep-hooking is the main driver in hooking mortality, our second study objective was to use angler observations noted above to evaluate deep hooking rates by Idaho Chinook Salmon and steelhead anglers using bait or other terminal tackle.

Methods

The Clearwater River, its tributary forks, the Little Salmon River, the Salmon River, and the Snake River are all popular steelhead and Chinook Salmon fisheries in Idaho. In these waters, anglers may only harvest steelhead with a clipped adipose fin; otherwise, they must release the fish. Anglers must also release Chinook Salmon with intact adipose fins except in rare occasions where returns are sufficient to allow harvest of unclipped fish.

We observed anglers fishing for steelhead and Chinook salmon in the main stem, South Fork, and North Fork of the Clearwater River, the Salmon River, and the Little Salmon River in Idaho. When possible, observations were made covertly, because we assumed that angler behavior might be affected by the close presence of state agency staff. However, many observations were overt, i.e., they were collected opportunistically during an unrelated program involving volunteer steelhead broodstock collection by anglers. Covert observations of anglers were conducted with binoculars from inconspicuous locations, or directly by observers posing as anglers.

When a fish was hooked, we used a stopwatch to measure the amount of time (s) it took from initial hookup to landing of the fish. At times the initial hookup was not observed so fight times were not available for some fish. Once landed, we timed how long the fish was exposed to air before being released. Occasionally, fish were put back in the water and

then re-exposed to air one or more times. As reported in Lamansky and Meyer (2016), re-exposures were relatively infrequent in the present study and did not meaningfully affect results. Thus, for the purposes of modeling analysis, we used the longest air exposure interval. During each fish landing event, we also noted various associated factors that might influence air exposure and fight times, including the method used to land the fish (net, hand), type of fishing gear used (fly, lure, bait), whether the angler was on foot or in a boat, and whether a photo was taken.

Non-fly fishing tackle such as beads, yarn, or bait drifted with or without a bobber were all fished very similarly and were not always distinguishable at a distance by anything other than the rod type and technique used, so they were combined into a non-fly gear category. Because rods used to fish lures are very similar to those used with other non-fly gear, and would be expected to have a similar effect on fight and air exposure times, lures were also included in the non-fly gear category. Thus, for fishing gear type used, we report either fly fishing or non-fly fishing when testing effects on fight time and air exposure. Water bodies were considered separately in this analysis due to differences among them that could contribute to variation in the data. For example, on the North Fork Clearwater River, many anglers fish either from the Ahsahka Bridge or from the wall below Dworshak Dam. At both locations, anglers are fishing 10-20 m above the surface of the water, which (1) precludes fly anglers from fishing those locations, and (2) greatly extends the fight time required to walk across the bridge or wall and climb down a series of stairs to reach the water and land a hooked fish. Thus, observations collected at the bridge and dam were treated separately when assessing fight times, but because this activity would not necessarily affect air exposure times, we did not separate this for air exposure data. In the interest of collecting independent observations, we tried not to collect more than one observation per angler for each day.

Hooking location was recorded for overt observations but could not be determined for covert observations; we assumed that anglers could not influence their hooking location based on their awareness of an observer, and we did not observe any snagging activities. Hooking location was recorded as deep (i.e., either in the gills or more deeply hooked), mouth (i.e., in the corner of the mouth or anything

inside the mouth but not deep hooked), or foul hooked. For hooking location, gear was categorized into either bait/jig, lure, or fly because of suspected differences between lures and bait (Muoneke and Childress 1994; Bartholomew and Bohnsack 2005).

The air exposure and fight time data represent time-to-event data that conformed to an exponential distribution, so we used accelerated failure time models to evaluate the factors affecting each response variable (Therneau and Grambsch 2000; Therneau 2015); air exposure and fight time were modeled separately. Accelerated failure time models designate a family of models that can be generalized to include covariates on the air exposure or fight-time function (Kalbfleisch and Prentice 2002). Models included water, gear type, photo taken (yes/no), observer status (covert/overt), and landing method (net/hand) as factors potentially affecting air exposure or fight time, which were ranked using Akaike's Information Criterion (AIC; Burnham and Anderson 2002). Once exponentiated, coefficients in the accelerated failure time models are multiplicative. For instance, if the coefficient for when a photo was taken was 1.5 for air exposure, this can be interpreted as meaning that air

exposure was 1.5 times or 50% higher for anglers who took a photo compared to those who did not. Program R was used for all data analysis (R Development Core Team 2011).

Results

From May 2016 to April 2017, we observed a total of 441 fish caught, from which 401 fight times and 259 air exposure times were recorded. Steelhead comprised 99% of the fight time observations and 97% of the air exposure observations. The longest interval of air exposure for caught-and-released salmon and steelhead averaged 28.7 s (± 3.2). The longest interval constituted 93% of the total air exposure that fish experienced because only 14% of anglers held fish out of water for two separate intervals and only 3% of anglers exposed fish to a third interval of air exposure. The average fight time was 131 s (± 10.6) (Table 1). Fly anglers photographed their catch 38% of the time, while non-fly anglers photographed their catch 25% of the time.

The model that included observer status, gear type, water, and whether a photo was taken best supported our air exposure time data (Table 2). Of these factors,

Table 1. Summary statistics of Chinook Salmon and steelhead air exposure and fight times including sample numbers (N), mean, range, standard deviation (SD) and 95% confidence intervals (CI) for gear types (fly and non-fly), observer status (covert and overt), and whether the angler photographed their catch. *Fight times for non-fly gear and for covert observations include fish caught from the south fork Clearwater and Little Salmon rivers only, as these were the only sites to have fly anglers and overt observations for comparison.

	N	Mean	Range	SD	95% CI
Air exposure (s)	259	28.7	0-185	26.3	3.2
Fly	49	23.5	0-89	21.9	6.1
Non-fly	210	29.9	0-185	27.2	3.7
Covert	152	32.3	0-185	31.3	5.0
Overt	103	23.2	2-60	14.5	2.8
Photo	78	38.6	0-129	29.0	6.4
No photo	180	24.4	0-185	24.0	3.5
Fight time (s)	401	131.0	5-900	108.7	10.6
Fly	69	169.8	13-765	133.8	31.6
Non-fly*	192	78.4	5-292	58.0	8.2
Covert*	47	74.16	5-494	91.8	26.2
Overt	171	113.2	6-765	101.8	15.3
Photo	103	157.2	17-519	108.2	20.9
No photo	296	121.7	5-900	107.7	12.3

Table 2. Akaike's Information Criteria (AIC) ranking for air exposure models.

Model	df	AICc	Δ AICc	weight
AirExp~Photo+Water+Covert+Gear	9	1859.8	0	0.961
AirExp~Photo	2	1867.2	7.33	0.025
AirExp~Water	6	1868.3	8.48	0.014
AirExp~Covert	2	1876.3	16.47	0
AirExp~Gear	2	1878.3	18.44	0
AirExp~Intercept	1	1878.4	18.59	0

Table 3. Coefficients, confidence intervals (CI), and *P*-values for the most highly supported air exposure model. (LS=Little Salmon River, NFB=North Fork Clearwater River dam wall and bridge fisheries, NFCLW= North Fork Clearwater River, SR=Salmon River, SFCLW=South Fork Clearwater River).

	Coefficient	95% CI	<i>P</i> -value
(Intercept)	17.14	6.91-42.52	<0.001
Photo Yes	1.53	1.13-2.07	0.006
Non-fly (Gear)	1.20	0.83-1.72	0.336
LS (Water)	0.39	0.17-0.92	0.030
NFB (Water)	0.97	0.43-2.21	0.948
NFCLW (Water)	0.67	0.25-1.77	0.419
SR (Water)	1.93	0.27-16.29	0.547
SFCLW (Water)	1.02	0.46-2.27	0.951
Covert Yes	1.58	1.09-2.29	0.015

photo taking had the strongest effect, with air exposure time being 1.53 times (~15 s) longer if the angler took a photo of their catch, after accounting for observer status, gear type, and water (Table 3). This effect appeared most evident for anglers using non-fly gear (Figure 1). The next most important factor affecting air exposure was observer status, with covert data collection resulting in 1.58 times longer air exposure. Other variation in air exposure was explained by differences in waterbody. Angler gear (fly vs non-fly) was not a significant explanatory variable for air exposure (Figure 2).

The model that included gear, water, observer status, and landing method best supported our fight time data (Table 4). The strongest predictor of fight time was gear, followed by observer status and

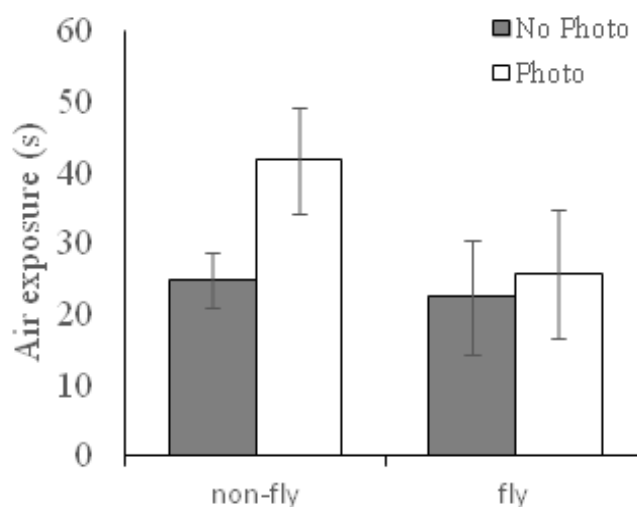


Figure 1. Average air exposure times (s) for Chinook Salmon and steelhead caught and released by non-fly and fly anglers who did or did not photograph their catch. Bars represent 95% confidence intervals.

waterbody (Table 5). When fly-gear was used, anglers fought fish for 1.57 times (76 s) longer than when non-fly gear was used (Figure 2). Fight time was also affected by observer status, but in the opposite direction as air exposure. Fight times measured covertly were 0.49 times (68 s) as long as those collected overtly. Differences in waterbody where data were collected also influenced fight time variation. Fight times from fish caught from the dam wall or bridge on the North Fork Clearwater were more than 2 times longer. However, fight times on the South Fork Clearwater were 0.45 times as long on average.

Deep hooking rates were 0% for bait/jig, 1% for lures, and 0% for flies (Table 6). Foul hooking rates were much higher for lures (40%) than for flies (8%) or bait/jig (4%).

Discussion

Our finding that most Idaho anglers exposed caught-and-released salmon and steelhead to < 30 s of air concurs with a previous study of Idaho trout anglers, which also showed that fish on average were exposed to < 30 s of air before being released (Lamansky and Meyer 2016). Our average fight times of 120 s are longer than their reported 53 s, a disparity most likely explained by the size of the fish being caught (i.e., anadromous salmonids are approximately 10 times heavier than resident trout). Although many

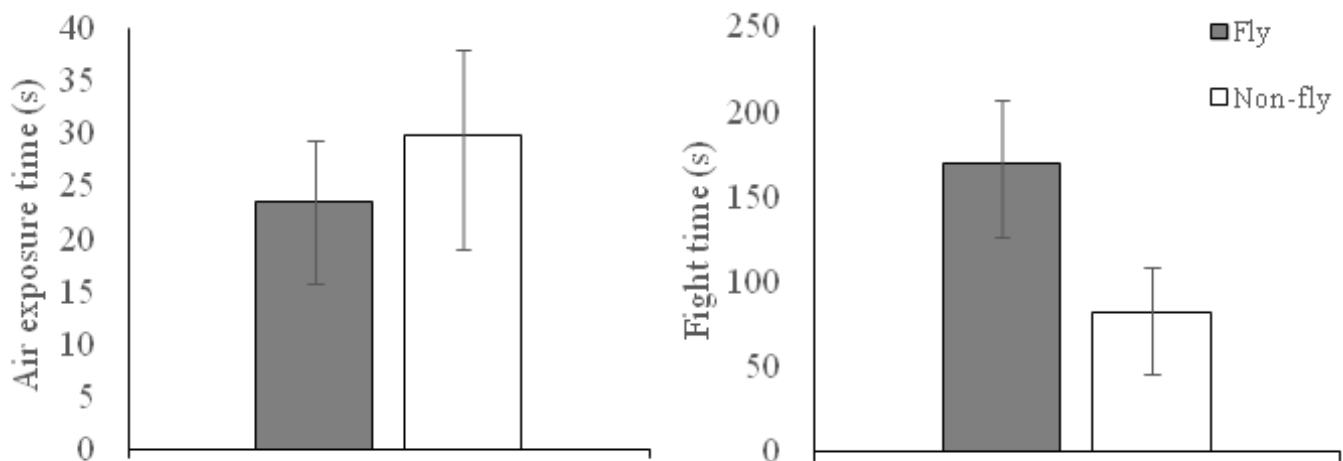


Figure 2. Average air exposure and fight times (s) for Chinook Salmon and steelhead caught and released by fly and non-fly anglers in Idaho. Bars represent 95% confidence intervals. Fight times are from the south fork Clearwater and Little Salmon rivers because these are the only sites that allowed for comparison of fly and non-fly anglers.

Table 4. Akaike's Information Criteria (AIC) ranking for fight time models.

Model	Df	AICc	ΔAICc	weight
Fight~Gear+Water+Covert+Landing method	9	4471.6	0	0.767
Fight~Photo+Gear+Water+Covert +Landing method+Harvest	11	4474	2.39	0.233
Fight~Water	6	4515.1	43.5	0
Fight~Harvest	2	4533.5	61.89	0
Fight~Gear	2	4535.9	64.27	0
Fight~Photo	2	4536.6	65.05	0
Fight~Covert	2	4537.8	66.2	0
Fight~Intercept	1	4540.2	68.65	0
Fight~Landing method	2	4542	70.37	0

Table 5. Coefficients, confidence intervals (CI), and P-values for the most highly supported fight time model. (LS=Little Salmon River, NFBD=North Fork Clearwater River dam wall and bridge fisheries, NFCLW= North Fork Clearwater River, SR=Salmon River, SFCLW=South Fork Clearwater River).

	Coefficient	95% CI	P-value
Intercept	454.89	216.35-956.43	<0.001
Gear (Non-fly)	0.43	0.32-0.57	<0.001
LS (Water)	0.63	0.31-1.24	0.177
NFBD (Water)	2.07	1.07-3.99	0.031
NFCLW (Water)	1.84	0.89-3.78	0.098
SR (Water)	0.68	0.09-5.40	0.714
SFCLW (Water)	0.45	0.23-0.88	0.020
Covert (Yes)	0.49	0.35-0.68	<0.001
Landing method (No net)	1.28	0.86-1.91	0.222

Table 6. Hooking locations for Chinook Salmon and steelhead caught on various gear types.

Hook location	Bait/Jig	Lure	Fly
Mouth	47	58	37
Foul	2	40	3
Deep (Gills/Esophagus)	0	1	0

physiology studies have simulated fight times for anadromous fish of up to 10 min or more, our results suggest that salmon and steelhead anglers in riverine settings land fish much more quickly. Minimizing fight times inherently reduces the cumulative stressful effects of handling by anglers. Until other studies demonstrate contrasting results, we suggest that future experimental research focus more closely on realistic fight times to evaluate the effect that exhaustive exercise may have on hooking mortality of caught-and-released fish.

The effects of air exposure and fight time on trout and salmon mortality has been tested in numerous studies (e.g., Ferguson and Tufts 1992; Schisler and Bergersen 1996; Schreer et al. 2005; Donaldson et al. 2014; Gale et al. 2014). In nearly all studies conducted to date, mortality rates are usually negligible when air exposure and fight times are representative of actual angler behaviors. The most cited study on fight and air exposure impacts to caught-and-released fish was conducted by Ferguson and Tufts (1992), who found mortality rates of 38% and 72% for hatchery Rainbow Trout exposed to only 30 and 60 s of air, respectively. In their study, test fish were chased for 10 minutes (simulating fight time), to the point of complete exhaustion for most fish, before exposing fish to air. Fish were then cannulated and repeatedly bled (five times) over the next several hours. Test conditions were so stressful in this experiment that even those fish not exposed to air suffered a mortality rate of 12%. For this reason, the authors explicitly stated that their results should not be applied to trout fisheries in the wild. Nonetheless, their results are repeatedly touted as demonstrating the severe negative impacts of small levels of air exposure on caught-and-released fish (Schisler and Bergersen 1996; Cooke and Suski 2005).

Our study confirms the importance of covertly collecting angler observational data so as not to bias

their behavior. As suspected, anglers did not hold fish out of the water as long when the data were overtly collected, presumably because they knew their behavior was being monitored. Previous studies that have reported angler fight and air exposure times have only included anglers participating in a particular study (e.g., Landsman et al. 2011). One surprising result was that covertly collected data had lower fight times than overt observations. This could be because most overt observations were collected opportunistically when biologists were cooperating with anglers that caught brood fish for spawning at a nearby hatchery. This situation may have placed greater importance on landing the fish, making the anglers more careful not to hurry the capture, thereby prolonging the process.

Deep hooking rates are important in the context of catch-and-release angling because for a variety of species (including resident and anadromous salmonids) it has been shown that (1) bait fishing results in higher rates of deep hooking, (2) deep hooking more often results in injury to critical internal organs, and (3) such injuries greatly increase the mortality rate for released fish (Muoneke and Childress 1994; Bartholomew and Bohnsack 2005). While we did not estimate hooking mortality, we observed deep hooking rates ($\leq 1\%$) that were lower than those reported in most other anadromous salmonid studies. For example, Chinook Salmon in the Willamette River experienced a 13% deep hooking rate, including rates of 15% for bait anglers and 8% for anglers using spinners (Lindsay et al. 1004). Chinook Salmon in the Yakima River experienced an 8% deep hooking rate, with 99% of anglers fishing with bait (Fritts et al. 2016). Other anadromous salmonid studies investigating hooking locations have included the tongue and roof of the mouth as vital areas during data collection (e.g., Bendock and Alexandersdottir 1993; Cowen et al. 2007) because they can result in bleeding. However, the goal in any catch-and-release fishery is to maximize the rate at which anglers hook fish in the jaw or mouth while minimizing gill and esophagus hooking. Our results suggest that for anadromous fisheries in Idaho, hooking in areas that can cause internal organ damage (i.e., gills and deeper) is negligible regardless of terminal tackle.

Often the most vocal anglers calling for restrictions on gear type or air exposure are fly anglers, who tout such restrictions as reducing handling stress for caught-and-released fish. In our study, fly anglers

took considerably longer to land salmon and steelhead, did not expose fish to less air, were more likely to hold a fish out of water for picture taking, and did not deep hook fish less often compared to other terminal tackle. These results suggest that fish caught and released by fly anglers in Idaho anadromous fisheries experience more stressful handling conditions than fish caught by non-fly anglers. This result should not be surprising given the physical differences between rod types. Fishing rods used with bait, bobbers, and lures typically have much greater resistance and strength than fly rods used in similar fisheries. However, it should be noted that these differences, though statistically significant, are likely not biologically meaningful in the context of increasing post-release mortality rates or population-level impacts.

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Effects of Air Exposure on Survival and Fitness of Yellowstone Cutthroat Trout

Curtis J. Roth¹, Daniel J. Schill², Michael C. Quist³, Brett High⁴, Matthew R. Campbell⁵, Ninh Vu⁶

¹Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Dr. MS 1141, Moscow, Idaho, USA 83844 Roth7618@vandals.uidaho.edu, (208 792-1181)

²Idaho Department of Fish and Game, 600 S. Walnut, Boise, Idaho, USA 83712

³U.S. Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Dr. MS 1141, Moscow, Idaho, USA 83844

⁴Idaho Department of Fish and Game, 4279 Commerce Circle, Idaho Falls, Idaho, USA 83401

⁵Idaho Department of Fish and Game, 1800 Trout Rd., Eagle, Idaho, USA 83616

⁶Pacific States Marine Fisheries Commission, 1800 Trout Rd., Eagle, Idaho, USA 83616

Abstract—In recent years, concerns have been raised regarding the practice of exposing fish to air during catch-and-release angling. The purpose of this study was to evaluate the effects of air exposure on survival and reproductive success of Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri*. Pre-spawn fish were sampled at a velocity-barrier weir. While the gills remained underwater, each fish was measured, tagged with a passive integrated transponder (PIT) tag, a tissue sample was taken, and then hooked through the lower jaw with a circle hook. Fish were then played to simulate angling. After angling, fish were randomly assigned an air exposure treatment of 0, 30, or 60 s. After treatment, fish were released upstream. Adult fish were sampled at the weir from May through July, 2016. In total, 1,519 pre-spawn adult fish were sampled. We detected 216 tagged fish out-migrating from Burns Creek to the South Fork Snake River (0 s, n = 57, 12%; 30 s, n = 71, 14%; 60 s, n = 88, 16%) after spawning from June through August 2016. Additionally, age-0 fish (n = 2,924) were collected from July through October 2016 and tissue samples were taken for parentage analysis so that the effects of the air exposure on fitness could be evaluated. Results of this study provide insight on how air exposure influences survival and fitness of Yellowstone Cutthroat Trout.

Introduction

Despite the success and popularity of catch-and-release regulations, some concerns have been raised regarding the practice. Exposing fish to air during release is one of the most high-profile of these concerns (Cook et al. 2015). Air exposure is believed to temporarily suppress gas transfer across the gills (Ferguson and Tufts 1992). A lack of gas transfer across the gills for a long period can lead to hypoxia and increased levels of carbon dioxide in the bloodstream. However, it is largely unknown how long fish can be exposed to air before long-term negative effects are observed. A few studies have attempted to address the question of how long a fish must be exposed to air before it experiences negative effects, but the results are highly variable. Most studies

have shown little to no effect on fish after 2 or even 3 min of air exposure (e.g., Schisler and Bergersen 1996; Schreer et al. 2005; Suski et al. 2007; Gale et al. 2011; Raby et al. 2013;), whereas one study reported negative effects after only seconds (Richard et al. 2013). A few studies have evaluated the direct effects of air exposure on mortality in a number of different species. Once again, the majority of studies have found little to no effect (e.g., Davis and Parker 2004; Suski et al. 2007; Rapp et al. 2014; Louison et al. 2016; Gagne et al. 2017). However, two studies have reported a seemingly large effect of air exposure on mortality (Ferguson and Tufts 1992; Graves et al. 2016). Despite the fact that the majority of studies have shown little to no effect, the most often cited study in the air exposure literature is Ferguson and Tufts (1992) and other studies showing no effect are largely ignored. Ferguson and Tufts (1992) reported that Rainbow Trout *Oncorhynchus mykiss* exposed to air for 60 s after an exercise event had a 72% mortality rate compared to a 12% mortality rate for fish that were only exercised. Results of Ferguson

and Tufts (1992) are often used to argue for limited air exposure in wild fisheries, but their individual study fish were cannulated and subject to five blood drawings and handlings. Appropriately, the authors note in the paper's discussion that their results are not applicable to wild fisheries. In addition to direct mortality, a small number of studies have evaluated the effects of air exposure on reproductive success with vastly different conclusions. For example, Atlantic Salmon *Salmo salar* in the Escoumins River, Quebec, exhibited decreased reproductive success when exposed to air (Richard et al. 2013). Fish exposed to air for more than 10 s had two to three times lower reproductive success compared to fish exposed to air for less than 10 s. Proponents of limiting air exposure often cite this decline in reproductive success as a reason to limit air exposure (e.g., Cook et al. 2015). Conversely, it has been reported that upon reaching their spawning grounds in Weaver Creek, British Columbia, Chum Salmon *Oncorhynchus keta* and Pink Salmon *O. gorbuscha* became resilient to any effects of catch-and-release angling (Raby et al. 2013). In fact, no decline in spawning success was reported after simulated capture and 1 min of air exposure. Given the variable and often contradictory results in the air exposure literature, further study on its effects on sport fish is warranted before regulations limiting the amount of time that anglers can expose fish to air during catch-and-release angling are seriously contemplated or implemented. Further, the artificial nature of many past air exposure studies (e.g., simulating angling via extended periods of tail-grabbing of hatchery trout in a raceway) calls for additional studies on wild fish under conditions similar to real world catch-and-release fishing.

Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* are an ideal species for evaluating the effects of air exposure because salmonids are among the most sensitive taxa with regards to hypoxia (Doudoroff and Shumway 1970) and support important recreational fisheries throughout the intermountain west (Quist and Hubert 2004). The objectives of the current study were to evaluate the effects of actual catch-and-release air exposure on survival and reproductive success of Yellowstone Cutthroat Trout.

Study Area

The current study was conducted in the South Fork Snake River (SFSR) drainage on Burns Creek,

from May through October, 2016. Burns Creek is a third-order tributary of the SFSR (Moore and Schill 1984). Discharge in Burns Creek varies from 0.1 to 9 m³/s and channel gradient is 3-6%. A large portion of the Yellowstone Cutthroat Trout population exhibits a fluvial life history, where fish move from the main stem SFSR into Burns Creek to spawn (Thurow et al. 1988). Yellowstone Cutthroat Trout in the SFSR mature around age 4; spawning begins in late May and continues through early July. Approximately 2 weeks after spawning, adults migrate from Burns Creek back to the main stem of the SFSR. Fry typically emerge from mid-July through September. Fry typically out-migrate at age 0 in Burns Creek (Moore and Schill 1984; Thurow et al. 1988).

Methods

Adult Yellowstone Cutthroat Trout were sampled at an existing IDFG velocity-barrier weir. Fish must enter a fish ladder to navigate the weir and continue upstream. While the gills remained underwater, a 12-mm full-duplex passive integrated transponder (PIT) tag was inserted into the peritoneal cavity of each fish (Prentice et al. 1990). Returning fish that already had a PIT tag were scanned to record the tag number and had a needle inserted into the peritoneal cavity to mimic a PIT tag injection. All newly PIT-tagged fish had their adipose fin removed as a secondary mark and the sample was retained for future individual genetic identification. Tissue samples were also taken from the upper caudal fin of fish lacking an adipose fin. Both tissue sample types were stored on Whatman 3MM chromatography paper (Thermo Fisher Scientific, Inc., Pittsburgh, Pennsylvania) for genetic analysis by the IDFG Genetics Lab. While remaining underwater, fish had a 1/0 circle hook manually inserted through the middle of their lower jaw and were randomly assigned a treatment of 0 (control), 30, or 60 s of air exposure. Fish were quickly maneuvered into a submerged 10.2-cm acrylic tube, measured for total length while remaining underwater, and carried upstream of the weir. Angling gear was attached to the circle hook in the fish's lower jaw and they were returned to the river and played. Fish were played for 102 s, the average landing time for spawning-sized Yellowstone Cutthroat Trout in a similar discharge, large river catch-and-release fishery on the Yellowstone River (Schill et al. 1986). After being played, fish were netted using a rubber-

mesh net, unhooked, exposed to their prescribed air exposure treatment, and returned to the river to move upstream and spawn. An estimate of relative survival by treatment group was then calculated using two fixed PIT-tag antennas located 0.5 km downstream of the weir to detect adult fish as they out-migrated back to the SFSR. Adults recorded by the PIT-tag antenna were assigned back to their air exposure treatments as an estimate of relative survival. Once adult trapping had concluded, collection of out-migrating age-0 fish began to evaluate the possible effect of adult pre-spawning air exposure on subsequent reproductive success. Fry collection was conducted using two trapping methods and electrofishing between July and October, 2016. A modified picket weir was installed downstream of the IDFG velocity-barrier and a Kray-Meekin trap was placed in the thalweg downstream of the picket weir. Single-pass backpack electrofishing was also conducted during a 2-d period in both September and November, 2016 to collect fry samples for parentage analysis (Richard et al. 2013). Burns Creek was sampled from the existing IDFG velocity-barrier weir upstream for 4 km, where the majority of fluvial fish spawn (Brett High, IDFG, personal communication). Once tissue samples were collected from all fry sampled with the three methods, DNA was extracted from all adult and fry tissue at the IDFG Eagle Genetics Lab in Eagle, Idaho. A suite of 141 single nucleotide polymorphisms (SNPs) were used to sequence adult DNA samples, and parentage-based tagging or PBT assignment was subsequently conducted to determine the parentage of each fry (Steele et al. 2013).

Data Analysis

The effects of air exposure on the survival of adult Yellowstone Cutthroat Trout caught and released during the spawning season was evaluated by calculating the proportion of fish from each treatment group that were detected moving downstream past the PIT-tag antenna located in lower Burns Creek. The proportions were statistically compared between groups by calculating confidence intervals around the differences between group proportions (Fleiss 1981; Johnson 1999). The differences between proportions were then considered statistically significant if the calculated confidence interval did not include zero. Evaluation of the effects of air exposure on the reproductive success of spawning Yellowstone

Cutthroat Trout was analyzed using generalized linear models with a negative binomial distribution. Two sets of candidate models were developed: one set using only data collected on adult male Yellowstone Cutthroat Trout (male-only models) and a second set using only data collected from adult female Yellowstone Cutthroat Trout (female-only models). A total of four candidate models was developed for both male-only models and female-only models. A priori models included a model using only fish length as a predictor variable, a model using only air exposure treatment as a predictor variable, a model using both air exposure treatment and fish length as predictor variables, and a model including air exposure treatment, length of the fish, and the interaction between fish length and air exposure treatment. Models were compared using Akaike Information Criterion (AIC) and the top model was the model that had the lowest AIC value (Burnham and Anderson 2002). Models that had an AIC score within 2.0 AIC values of the best model were also considered top models.

Results

In total, 1,518 upstream migrating adult fish were sampled and assigned to a treatment group (0 s, $n = 487$; 30 s, $n = 495$; 60 s, $n = 536$). Downstream migrating adult fish (post-spawners) were detected at two PIT-tag antennas located downstream of the weir as they out-migrated to the SFSR. We detected a total of 216 tagged fish migrating downstream and relative survival ranged from 12% in the 0 s treatment group to 16% in the 60 s treatment group (0 s, $n = 57$, 12%; 30 s, $n = 71$, 14%; 60 s, $n = 88$, 16%). Confidence bounds around the difference between proportions did overlap zero, indicating that estimated relative survival was not different between fish treated with 0 s or 30 s or between fish treated with 30 s or 60 s exposure. However, estimated relative survival was significantly different between fish treated with 0 s of air exposure and fish treated with 60 s of air exposure. Unexpectedly, fish exposed to air for 60 s had a slightly higher estimated relative survival rate than both the 0 and 30 s treatment groups. We sampled 2,924 fry (electrofishing, $n = 2,175$; Kray-Meekin, $n = 583$; picket weir $n = 166$). All 2,924 fry were successfully genotyped and of those 2,310 were assigned back to two parents that were given an air exposure treatment at the weir on Burns Creek. The top male AIC model only contained air exposure treatment and fish length as predictors. However, air exposure

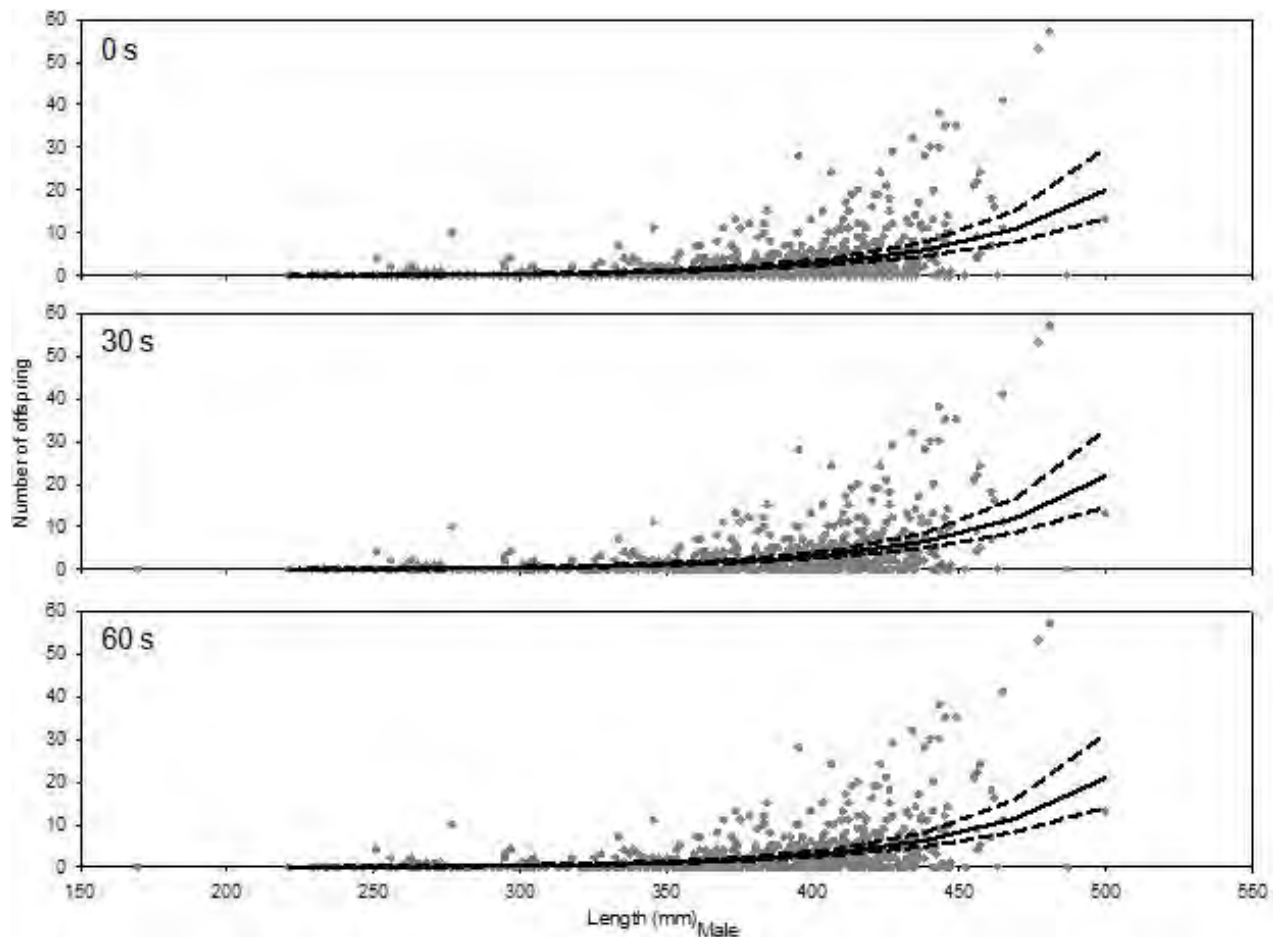


Figure 1. Scatter plot representing the number of offspring produced by adult male Yellowstone Cutthroat Trout of various lengths in Burns Creek, Idaho. Fish were then subjected to one of three air exposure treatments (i.e., 0, 30, or 60 s) and played on hook-and-line gear to simulate angling. Parentage was determined by genetic analyses of the adults and fry. The solid line represents the predicted number of offspring produced based on a generalized linear model using air exposure treatment and the length of the fish to predict the number of offspring a fish would produce at a given length. The dotted line represents a 95% confidence interval on the estimated number of fry. Sampling of adults and fry occurred between May and October, 2016.

treatment did not significantly affect the number of offspring produced in the male-only models (30 s, $Z = 0.562$, $P = 0.574$; 60 s, $Z = 0.270$, $P = 0.367$; Figure 1). Only fish length significantly affected the number of offspring produced ($Z = 12.232$, $P < 0.001$; Figure 1). Results of regression analysis for the female-only models were similar to those of male-only models with the top model containing only the predictor variables of air exposure treatment and fish length. The top female model indicated that air exposure treatment did not significantly influence the number of offspring produced (30 s, $Z = -0.787$, $P = 0.431$; 60 s, $Z = -0.903$, $P = 0.367$; Figure 1), but the length of the

fish did significantly affect the number of offspring produced ($Z = 7.642$, $P < 0.001$; Figure 1). Thus, air exposure treatment did not influence reproductive success, however adult length did significantly influence offspring production.

Discussion

No negative effect of air exposure at the two treatment levels examined (30 and 60 s) were observed in terms of mortality of pre-spawn adult Yellowstone Cutthroat Trout. The vast majority of previous studies evaluating the effect of air exposure on mortality

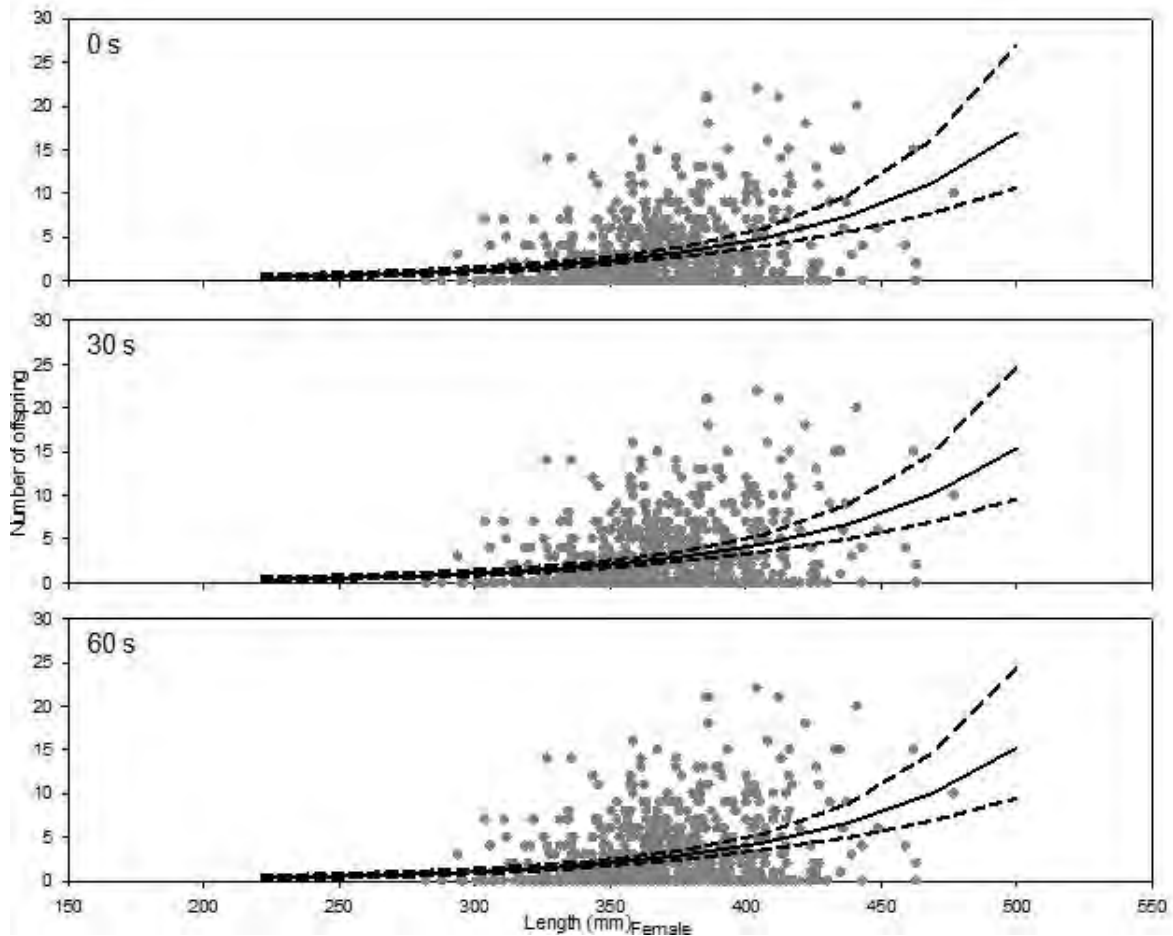


Figure 2. Scatter plot representing the number of offspring produced by female Yellowstone Cutthroat Trout of various lengths in Burns Creek, Idaho. Fish were then subjected to one of three air exposure treatments (i.e., 0, 30, or 60 s) and played on hook-and-line gear to simulate angling. Parentage was determined by genetic analyses of the adults and fry. The solid line represents the predicted number of offspring produced based on a generalized linear model using air exposure treatment and the length of the fish to predict the number of offspring a fish would produce at a given length. The dotted line represents a 95% confidence interval on the estimated number of fry. Sampling of adults and fry occurred between May and October, 2016.

have also reported that when air exposure times were similar to those used in the current study air exposure had little to no negative effect on mortality (e.g., Schreer et al. 2005; Gingerich et al. 2007; Suski et al. 2007; Thompson et al. 2008). For example, Brook Trout *Salvelinus fontinalis* were exposed to air in a laboratory for either 0, 30, 60, or 120 s and no mortality was reported (Schreer et al. 2005). As mentioned previously, the majority of studies that have been conducted to evaluate the effect of air exposure on mortality have had a limited applicability to wild fish populations and therefore limit the ability of fisheries managers to make informed decisions. In

fact, only a handful of studies have dealt with wild un-caged fish populations, and with one exception (i.e., Graves et al. 2016) they have largely found no effect on mortality (i.e., Thompson et al. 2008; Louison et al. 2016; Gagne et al. 2017). For example, wild Northern Pike *Esox lucius* from Grand Lake, Wisconsin, were exposed to air for up to 4 min during ice angling and they exhibited no immediate mortality (Louison et al. 2016). The one exception to studies showing that air exposure had little to no effect on mortality in wild fish populations reported that White Marlin *Kajikia albigata* captured off the coast of Virginia Beach, Virginia, via angling had a 16.7% rate of

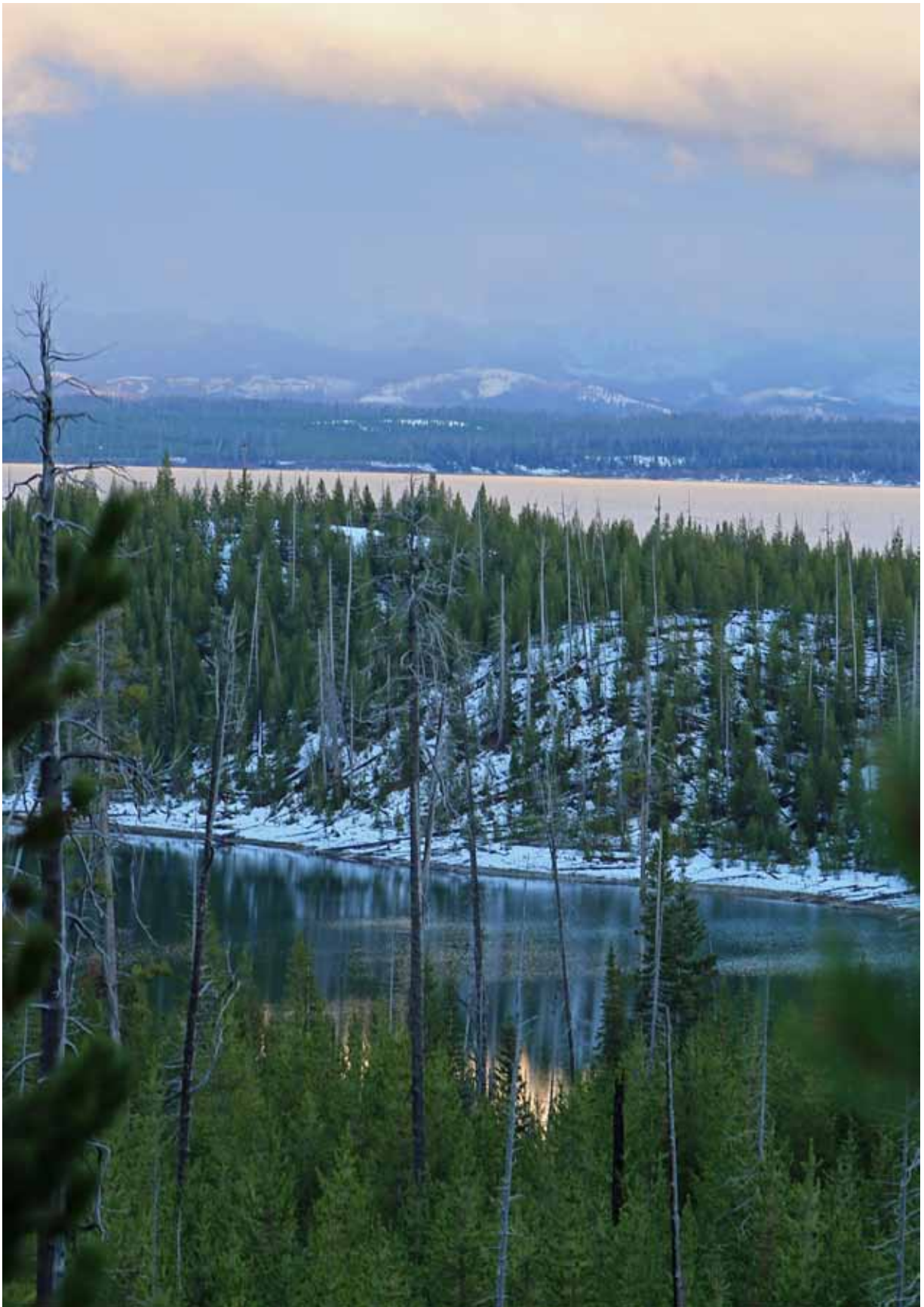
mortality when exposed to air for 1 min compared to a 1.7% rate of mortality of fish that were not air exposed (Graves et al. 2016). However, results of Graves et al. (2016) must be interpreted with caution for several reasons. Perhaps the most notable being the small sample size of fish in each treatment (i.e., 1 min, n = 6; 3 min, n = 5; 5 min, n = 7). Secondly, the control fish used to evaluate the effects of air exposure treatments were fish captured 8 years prior in a different study (Graves et al. 2008) and were caught largely in other locations. A single study has reported a decline in reproductive success due to catch-and-release air exposure. Atlantic Salmon *Salmo salar* in the Escoumins River, Quebec, exhibited decreased reproductive success when exposed to air (Richard et al. 2013). Fish exposed to air for more than 10 s had two to three times lower reproductive success compared to fish exposed to air for less than 10 s. However, the Richard et al. (2013) study suffers from the same sample size issues as noted above as only 40 caught-and-released adult Atlantic Salmon were involved in the entire study with only 24 being caught, exposed to air, and released. Additionally, the author's model indicated that increased air exposure resulted in increased reproductive success when the water was warm (> 17°C), a result at odds with virtually all prior studies. Conversely, no decline in spawning success was reported after simulated capture and air exposure of up to 60 s for Chum Salmon and Pink Salmon in Weaver Creek, British Columbia (Raby et al. 2013).

When results of the current study are combined with the results of prior air exposure studies it seems unlikely that increased mortality due to air exposure from catch-and-release angling would be a concern in the vast majority of catch-and-release fisheries. Further support is brought to this view when considering the length of time anglers exposed to air in actual catch-and-release fisheries was only 29 s (Lamansky and Meyer 2016). Therefore real observed air exposure times in real world catch-and-release fisheries are far less than those times used in the prior studies that looked at the effects of air exposure on mortality in wild fish populations (i.e., Thompson et al. 2008; Louison et al. 2016; Gagne et al. 2017). We conclude that natural resource agencies should exercise caution when considering regulations that place limitations on air exposure in catch-and-release fisheries.

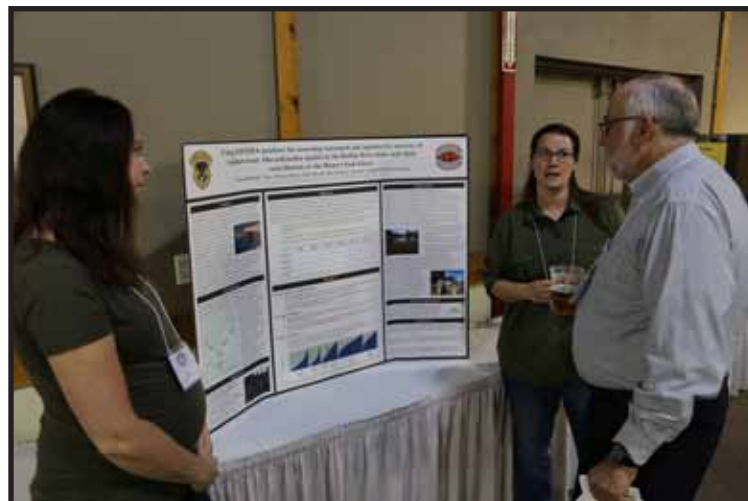
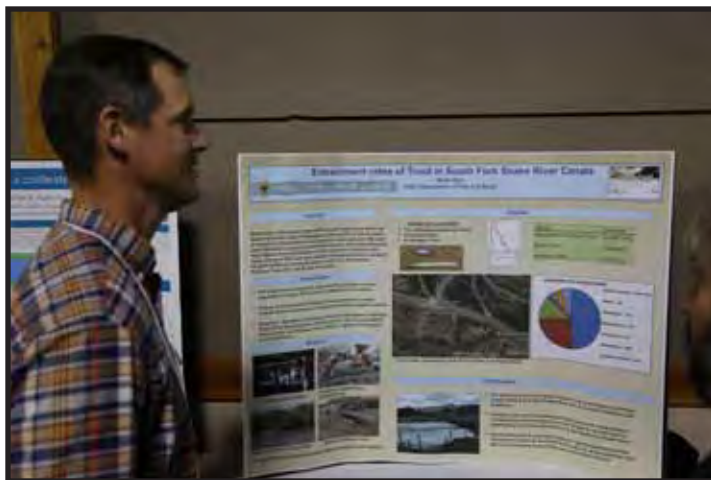
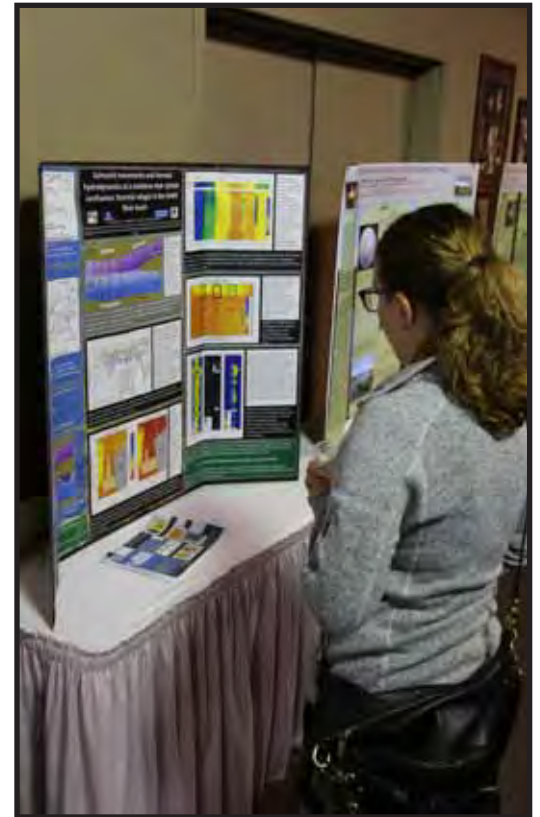
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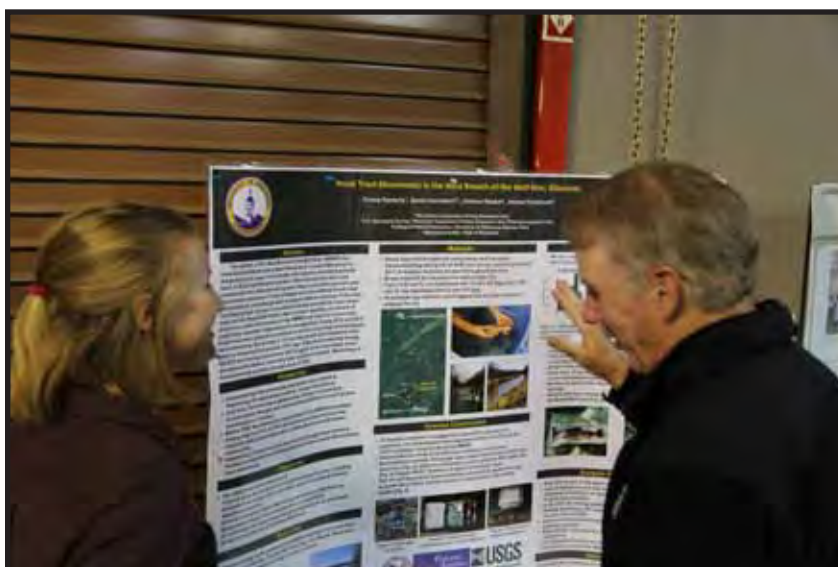
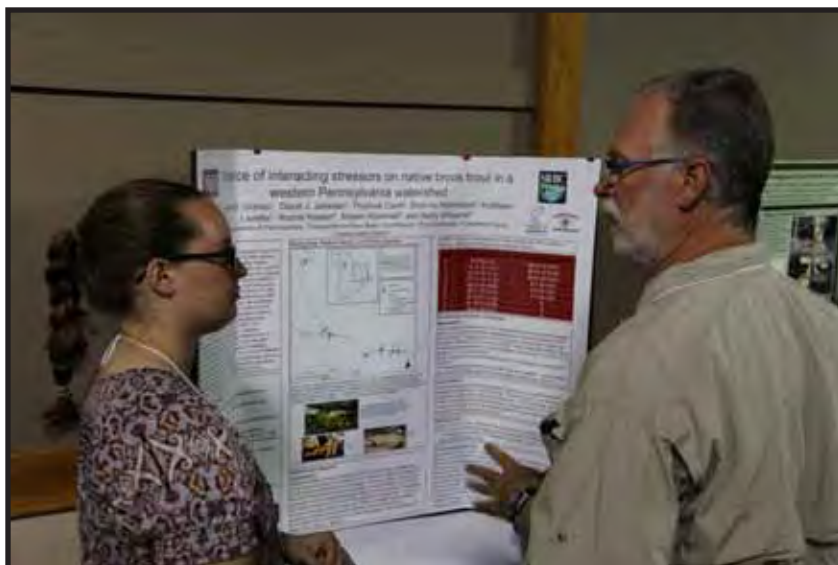
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Poster Presentation





Poster Presentation Abstracts (in alphabetical order)

Presenting Author	Poster title and abstract
Ayers, Paul	Kayak-Based Underwater Videomapping System for Watershed-Scale Wild Trout Habitat Management Abstract: The need to develop GIS-compatible large-scale maps of aquatic habitat in river systems led to the design of a kayak-mounted GPS-based river videomapping system. The river mapping system is kayak-mounted with georeferenced above and under water cameras, depth sounder, width sensors, and underwater lasers. GIS maps of river and streambank characteristics - (pool, riffle, run), substrate (modified Wentworth scale), embeddedness (EPA classification), woody debris, bank cover, depth, width and river characteristic (pool, riffle, run) were developed. River thalweg rugosity and sinuosity were also determined. Every linear foot of river can be mapped at a rate of 10 miles per day. The system provides a GIS-based georeferenced database for river and stream inventory and can be used for watershed-scale wild trout habitat management. A technique to define optimum habitat locations and habitat suitability indices for aquatic species was developed and implemented. Complemented with a GPS-based snorkel videomapping system (GSVMS) and a Sneak Peek under-structure video exploration technique, site-specific fish population monitoring provide video documented georeferenced information regarding population, size, species distribution, location, and habitat. GIS-based video tours of the above and below water river features, providing virtual tours within ArcGIS and Google Earth will be demonstrated.
Ayers, Paul	Wild trout Population Surveys Using Day and Night GPS-based Snorkel videomapping in the Great Smoky Mountains and Yellowstone National Park Abstract: Populations were mapped and populated habitats were identified: deep or shallow depths; boulder, cobble or sand substrate; and pool/riffle/run river characteristics. Data for the snorkeling effort - time in water, distance covered, and locations snorkeled with ArcGIS reference points were also documented. Using mask-mounted underwater lights, the snorkel videomapping wild trout population counts were conducted during the day and night. Comparison of fish populations revealed a substantially different population distribution. The GSVMS also provides an opportunity for volunteers to conduct a “rapid bioassessment inventory,” while the video can be reviewed later by fisheries biologists.
Dotts, Sandy	Low-Tech Success: Large Wood Replenishment in Western Trout Streams Abstract: For most western trout stream systems, large wood is the key to providing good fish habitat. However, past management practices have left many streams severely lacking wood. Typical large wood debris (LWD) restoration projects include engineered designs, off-site logs, heavy equipment installation, and lots of money. Structures were designed and installed to never move. We now understand that streams are dynamic and that in natural systems, some wood moves and this movement creates more complex and stable habitat. Efforts to “control” LWD is simply a tired, ineffective and cost prohibitive approach to restoration. We have successfully used a low-tech, cost effective, and low-impact method to increase the amount and distribution of LWD in Bull Trout <i>Salvelinus confluentus</i> and Westslope Cutthroat Trout <i>Oncorhynchus clarkii lewisi</i> streams. Using mostly hand labor and trees from the adjacent riparian zone, we have installed nearly 600 logs/log complexes in over 13 miles of stream since 2011. Trees were directionally-felled into stream channels at strategic locations to increase cover, resting and high flow habitat, and channel complexity. In addition to single logs, felled trees were also moved using hoists to create multi-log structures. Passive anchoring (trees on top of other trees or braced against boulders/rootwads) was used to limit movement within the stream and protect downstream infrastructure. No cables were used to prevent log movement and no heavy equipment entered the streams or riparian zones. Construction cost was about \$200/log or about 1/5th the cost of traditional log placement methods using heavy equipment. High-resolution, low elevation images of the streams were taken before and after installation to monitor structure movement and physical stream responses. Monitoring has found that, as predicated, some wood moved during high spring runoff events, including record high flows. Overall, this low-tech method has proved highly successful in improving habitat condition for western trout.

Presenting Author	Poster title and abstract
Easterly, Emma	Brook Trout Movements in the West Branch of the Wolf River, Wisconsin Abstract: The portion of the West Branch of the Wolf River (WBWR) that resides within the Menominee Indian Reservation is classified as a class I trout stream with naturally-reproducing Brook Trout <i>Salvelinus fontinalis</i>. Little is known regarding movement of Brook Trout within this section of the WBWR and seasonal movements could have important implications for management. Additionally, in 2015, the Menominee Indian Tribe removed two dams and constructed new channels with graded steps to promote upstream and downstream movement. These changes in stream connectivity may have affected Brook Trout movements by providing greater access to different portions of the river. Moreover, there may be more access to lacustrine habitats provided by Upper Bass Lake and the Neopit Mill Pond. The objectives of this study are to determine if (1) Brook Trout use multiple segments of the WBWR during the year, including sections of the stream where dams were removed and channel alterations occurred; (2) a large percentage of Brook Trout enter Upper Bass Lake and the Neopit Mill Pond as refuge from temperature change, and (3) Brook Trout can move freely up and down a rapids located just upstream of the Neopit Mill Pond. Brook trout will be captured at various transects using barge and backpack electroshocking during 2016 and 2017. Brook Trout ≥ 120 mm will be implanted with passive integrated transponder (PIT) tags and antenna deployments at multiple locations will be used to detect and describe trout movements. Observed movements during the fall of 2016 and summer of 2017 suggest Brook Trout freely move throughout the stream. Information obtained from the project will help tribal biologists in managing the Brook Trout population.
Fopma, Seth J.	Brown Trout Diet Analysis for Black Hills Streams: Implications for a Mountain Sucker Abstract: Brown Trout <i>Salmo trutta</i> undergo an ontogenetic diet shift early in life switching from a primarily insectivorous feeding strategy to piscivory. Piscivorous trout often live longer and exhibit increased growth rates when compared to trout with primarily insectivorous diets. Brown Trout were introduced into Black Hills streams in the 1930's to provide increased angler opportunities and satisfaction. Introduced nonnative fish can have detrimental effects on native fish populations. Mountain Sucker <i>Catostomus platyrhynchus</i> a species of conservation concern in South Dakota. Recent studies suggest that Mountain Sucker may experience localized declines due to potential displacement and piscivory from co-occurring Brown Trout. Fish were sampled during August and October of 2015 in two different watersheds across the Black Hills reflecting differing densities of Brown Trout and Mountain Sucker. All fish were collected, weighed, and measured (TL) before release. We performed gastric lavage (79 % efficiency) on Brown Trout greater than 200 mm (TL) ($n=192$) to examine diet and all trout (>120 mm) were tagged with a 12-mm half-duplex PIT tag prior to release. Fish contribution to diets was highest within the Whitewood Creek watershed (27%) during the summer. Mountain Sucker accounted for 8% of prey fish consumed, (Brown Trout = 20%, Longnose Dace, <i>Rhinichthys cataractae</i>, = 28% unidentified fish = 44%). Results suggest that while predation potential may be high, incidence is low, which may be due to rich invertebrate communities. Understanding these species interactions will aid in the identification and development of conservation and refuge areas needed by this imperiled species.
Graham, Christy	Growth of Brown Trout in the Greers Ferry Tailwater, Arkansas Abstract: In 2007, a 406-610 mm TL protected slot limit was enacted for the Greers Ferry Tailwater (GFTW) in an attempt to improve the overall size distribution of the trout population. However, the abundances of trout within and above the slot limit did not improve following enactment of the regulation. In order to evaluate the efficacy of the regulation, the Arkansas Game and Fish Commission (AGFC) initiated a mark-recapture study in January 2013 to examine growth rates of wild Brown Trout <i>Salmo trutta</i> in the fishery. Brown Trout were sampled via electrofishing in the upper 5.4 km of GFTW. On the first night of each sampling event, all Brown Trout collected were measured, weighed, and double-tagged with passive integrated transponder (PIT) and T-bar anchor tags. On two subsequent nights, all Brown Trout were checked for the presence of tags and untagged individuals were marked. A total of 209 Brown Trout (average 386 mm TL) were marked during the study; most fish tagged (68%) were 254-405 mm TL. A total of 117 marked Brown Trout were recaptured between March 2013 and July 2014. During that time, average growth rates of individuals ranged from 0 to 30 mm/year. In

Presenting Author	Poster title and abstract
	general, growth rates of fish decreased as fish length increased. Individuals that were between 250-275 mm TL when tagged grew at an average rate of 69 mm/year, whereas fish that were between 450-475 mm TL when tagged grew at an average rate of less than 1 mm/year. It is likely that the slow growth rates observed during this study contributed to the ineffectiveness of the protected slot limit. These data, in combination with AGFC electrofishing and creel data indicating high Brown Trout abundances and low angler harvest rates, supported AGFC's decision to remove the protected slot limit regulation in January 2017.
Graves, Jennifer	Native Brook Trout Responses to Interacting Stressors in a Western Pennsylvania Watershed Abstract: Freshwater species have declined throughout their native ranges in part due to habitat fragmentation and invasive species. Information is often lacking, however, about how interactions between these stressors affect certain aspects of native populations. Brook trout <i>Salvelinus fontinalis</i> are a prime example of a species in decline due to human-related stressors, two of which are fragmentation from abandoned mine drainage (AMD) and competition with nonnative brown trout <i>Salmo trutta</i>. In an ongoing, multi-year study, we are assessing the abundance, behavior, and genetic structure of Brook Trout in a Pennsylvania watershed fragmented by AMD and scheduled for remediation in 2018. Results from stream surveys show Brown Trout are absent upstream of AMD but abundant downstream, suggesting that AMD is a chemical barrier to Brown Trout invasion. This watershed represents a common situation in Pennsylvania—Brook Trout populations are simultaneously fragmented and “protected” from Brown Trout invasion by AMD, but remediation could permit Brown Trout invasion upstream. This balance between isolation and invasion presents a significant management challenge, and our study will help biologists predict likely outcomes under different management scenarios.
Griffin, Joanna	Using New Brook Trout Genetics Research to Improve Wisconsin's Trout Stocking Program and to Protect Wild Brook Trout Populations Abstract: Wisconsin Department of Natural Resources (WDNR) created a “wild” trout stocking program in 1995 to improve the genetics of trout stocked to restore wild populations. This program has evolved over the years by utilizing advances in genetics research to improve our broodstock selection and to protect the genetic integrity of wild Brook Trout <i>Salvelinus fontinalis</i> populations where they exist. In 2016 the WDNR began a new Brook Trout genetics study in collaboration with the University of Wisconsin-Stevens Point using data from multiple states in the Midwest and hatchery sources from the East Coast to better define and understand Brook Trout population genetics throughout Wisconsin. Results of the new genetic analyses indicated certain Brook Trout populations, which were thought to have genetics representative of wild Brook Trout, actually have genetics indicative of domestication. Using the new genetics research on Wisconsin Brook Trout, our objectives for improving Wisconsin's trout stocking program and for protecting wild Brook Trout populations include (1) update definitions of genetic stock boundaries by expanding our collection of Brook Trout genetics samples, (2) update guidance on broodstock collection and hatchery propagation practices, including the identification of new broodstock sources, and (3) update guidance on trout stocking practices based on genetics and watershed boundaries.
Haglund, Justin	Age Validation of Brown Trout in Driftless Area Streams in Wisconsin using Otoliths Abstract: Accurate and precise age estimation is crucial to understand fish population dynamics and manage fish populations. In this study, we used otoliths to validate the absolute age of Brown Trout <i>Salmo trutta</i> in four Driftless Area streams of southwestern Wisconsin. Age-1 Brown Trout were tagged with coded wire tags during spring in 2010-2015. Recaptured known-age trout were later collected for ageing. Otoliths and coded wire tags were extracted from 249 Brown Trout ages 1-5 and aged by three independent readers. Complete agreement (agreement by all readers) was 74%, partial agreement (agreement between at least two readers) was 98%, and consensus agreement (agreed upon age by all readers) with known age was 93% with a coefficient of variation of 9.4%. Consensus agreement by age varied from 81% for age 3 (n=31) to 98% for age 1 (n=132) and 100% for age 5 (n=6). We conclude that otoliths provide an accurate method for ageing Brown Trout in Wisconsin's Driftless Area streams and results from this study may guide future Brown Trout ageing efforts in other productive inland riverine systems.

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High, Brett	Entrainment of Trout in Canals in the South Fork Snake River Abstract: The South Fork Snake River (SFSR) supports an ecologically and economically significant population of native Yellowstone Cutthroat Trout <i>Oncorhynchus clarkii bouvieri</i> (YCT). The biggest threat to native YCT in the South Fork Snake River is nonnative Rainbow Trout <i>O. mykiss</i> (RBT) due to hybridization and competition. However, other threats do exist to the YCT population such as entrainment into unscreened irrigation diversions. There are seven large unscreened canals that divert substantial amounts (up to 7,000 cfs) of water from the SFSR. Most of these canals are in the middle portion of the South Fork and densities of YCT in the upper portion of the river are consistently higher than in the lower portion. The SFSR is a well-connected system with migratory or fluvial life history forms of YCT present. Where migratory YCT in the lower river must move past several diversions to access spawning tributaries, managers were concerned that entrainment was negatively affecting YCT abundance. We used radio-telemetry to assess entrainment of YCT, Brown Trout <i>Salmo trutta</i> (BNT) and RBT into canals in the SFSR drainage. From 2013 through 2015, we marked 929 trout with radio tags, including 731 YCT, 141 BNT, and 57 RBT. The movements of marked fish were monitored through 2016. We found entrainment into canals occurred each year. In each year entrainment occurred primarily in early summer. Overall entrainment rates averaged 12%. Yellowstone Cutthroat Trout had the highest entrainment rate at 14% followed by RBT (13%) and BNT (6%). Although entrainment of adult trout of the SFSR into canals does occur, impacts vary by species. Overall, the impacts of entrainment on abundances of adult trout species in the SFSR is likely low enough that compensatory mortality may ameliorate most of its effects.
Kientz, Jeremy L.	Survival, Abundance, Growth, and Movement of Wild Rainbow Trout in the Deerfield Reservoir System, South Dakota Abstract: The Deerfield Reservoir system in the Black Hills of western South Dakota is primarily managed as a put-and-take fishery through annual stockings of 12,000 catchable-sized Rainbow Trout <i>Oncorhynchus mykiss</i>; however, recent estimates show that natural reproduction in tributary streams and subsequent recruitment may account for up to 25% of the Rainbow Trout population. While an elimination or reduction of hatchery stockings has been considered, a lack of information regarding factors such as movement and emigration patterns, relative abundance, and apparent survival of wild Rainbow Trout has generated a need for additional research in order to guide future management decisions. A total of 380 Rainbow Trout were tagged with 12-mm PIT tags throughout 15, randomly selected 100-meter sites within two main tributaries (Castle Creek and South Fork Castle Creek) to the reservoir. Movement was monitored using fish recaptured in backpack electrofishing events, while emigration into Deerfield Reservoir was monitored using a passive in-stream reader near the mouth of the reservoir. Locations of recaptured trout (n=81) revealed minimal within and among stream movements. Only 3 of 81 fish (4%) were recaptured outside of the 100-m site in which they were tagged. Another 73 unique fish were detected by the passive in-stream reader emigrating from tributary streams into Deerfield Reservoir. Mean abundance estimates of wild Rainbow Trout were as high as 610 fish/ha (>90 mm TL) and 195 fish/ha (<90 mm TL), and were significantly greater in South Fork Castle Creek than in Castle Creek. A von Bertalanffy growth model for wild Rainbow Trout showed that growth of fish up to age 4 was relatively slow in comparison to other populations, reaching only 210 mm by age 4. Survival of wild Rainbow Trout in the Deerfield Reservoir system was estimated to be as low as 3% during the first year of life; however, survival increased each year up to 66% by age 4.
Miller, Diana	A Tale of Two Lakes Abstract: Superficially, Yellowstone (YL) and Jackson (JL) lakes appear to have much in common, including populations of native Yellowstone Cutthroat Trout <i>Oncorhynchus clarkii bouvieri</i> and nonnative Lake Trout <i>Salvelinus namaycush</i>; however, fisheries in these lakes are managed quite differently. Historic use, ecology, and differing management objectives all contribute to these differences. Yellowstone became the first national park 18 years before Wyoming achieved statehood. Conversely, Grand Teton National Park was designated after State management of fish and wildlife had been long standing. Yellowstone National Park first documented Lake Trout in YL in 1994. Shortly

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	after this discovery, the National Park Service recorded a decline in Yellowstone Cutthroat Trout numbers and adopted an aggressive Lake Trout removal program. Lake Trout were stocked in Lewis Lake in 1889 and quickly established a population downstream in JL. By the time suppression of Lake Trout was considered imperative in YL, the population in JL was well established and considered a valuable sport fish. Forage fish conditions in these lakes differ vastly. In YL cutthroat trout are the primary option. In JL, five options exist with cutthroat trout vastly outnumbered by other species. National Park Service policies, aimed at protecting native species, give priority to managing nonnative species where prudent and successful control is reasonably likely.
Reeser, Stephen J.	Observations of Changing Sympatric Wild Rainbow and Brown Trout Population in a Virginia Tailwater Abstract: An 11-km coldwater tailrace was created after the construction of Gathright Dam on the Jackson River (western Virginia) in 1980. Stocking of hatchery Rainbow Trout <i>Oncorhynchus mykiss</i> and Brown Trout <i>Salmo trutta</i> was discontinued after eight years (1989-97) once sustainable levels of natural reproduction were documented for both species. We monitored population dynamics of the developing wild Rainbow Trout and Brown Trout fishery over an 18-year period (1999-2016). Recruitment, with natural variation between years, remained consistent for both species over the study period. Relative abundance (fish/h of boat electrofishing) of adult (>120mm) Rainbow Trout ($r^2=0.88$; $p<0.05$) and Brown Trout ($r^2=0.70$; $p<0.05$) steadily increased 1999-2010. Relative abundance of 305-407 mm trout (both species) also steadily increased over the same period (RBT- $r^2=0.67$; $p<0.05$ / BNT - $r^2=0.70$; $p<0.05$). In 2009 fishing regulations were changed from a 305-mm minimum size limit for both species to a more protective 305-407 mm protected slot for Rainbow Trout and a 507 mm minimum size limit for Brown Trout. Relative abundance of Brown Trout 305-407 mm and >407 mm increased after the regulation change (ANOVA $p<0.05$). However, relative abundance of Rainbow Trout 305-407 mm decreased post-regulation change (ANOVA $p=0.03$). In addition, the proportion of Brown Trout in the overall trout population increased by 50% over the entire survey period. We documented significant increases in growth from 2006 to 2016 for both Rainbow Trout and Brown Trout based on comparison of von Bertalanffy growth curves ($P<0.05$). An angler-creel survey conducted in 2007 (prior to regulation change) revealed that 60% of total angler effort was directed specifically toward Brown Trout and anglers released 96% of all trout caught. It is unclear how and to what degree the new size limit regulation, interspecific competition, or abiotic factors may have influenced recent changes in the wild trout population. However, spawning success (CPUE of age 0 fish) is instrumental in influencing the relative abundance of adult trout.
Rehm, Travis R.	Brown Trout Movement in Response to Large Scale Density Reductions Abstract: Density-dependent growth in stream-dwelling Brown Trout <i>Salmo trutta</i> has been well documented in the literature. In Spearfish Creek, South Dakota, mean size of Brown Trout is inversely related to trout density. Due to low angler harvest, management options for increasing the size structure of the trout population may be limited. Experimental manipulation of Brown Trout abundance shows promise as an approach for increasing growth rate of stream-dwelling trout. While studies have shown that Brown Trout generally exhibit small home range sizes and strong site fidelity, the effect of reduced density on movement patterns in Brown Trout is poorly understood. In order for experimental reductions of fish density to have positive effects on fish growth, it is important that immigration from high to low density areas is negligible. To evaluate the influence of fish density on movement, we implanted a total of 18 Brown Trout with radio transmitters in Spearfish Creek. Seven experimental sections (425 m/section) were randomly selected to reduce brown trout density by 50% and seven sections (425 m/section) served as control reaches. Fish locations (GPS coordinates) were recorded during daylight hours from August 17, 2016 to January 30, 2017, and used to compare movement in experimental and control sections. Mean fish movement was similar in experimental (26.6 m, $n=10$, SE 4.0) and control (23.4 m, $n=8$, SE 3.0) sections. Similarly, gross movement was not statistically different between treatments at 882.7 m (SE 116.6) or 812.4 m (SE 175.8) for experimental and control sections, respectively. Based on telemetry data, it appears that immigration of Brown Trout into large scale experimental sections would be negligible. Given the mean distance moved between tracking dates, only fish occurring in the lower or upper 25 m of sections adjacent to an

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	experimental section would be expected to move into areas with reduced fish density – constituting only about 11% (50/425) of the reach.
Reynolds, Jonathan	Wildfire Devastates Great Basin National Park Bonneville Cutthroat Trout Stream Abstract: In the summer of 2000, Strawberry Creek became the first stream in Great Basin National Park (GRBA) to be chemically treated in preparation for Bonneville Cutthroat Trout <i>Oncorhynchus clarkii utah</i> (BCT) restoration. In the time between the BCT reintroduction in 2002 and the population survey in 2011, Strawberry Creek became the park's most productive BCT stream. BCT were present in 8 of the 9 survey sites, had an average population density of 755 fish/mile, and a maximum density of 1,867 fish/mile. In the summer of 2016 a wildfire burned through the majority of the watershed, and it was feared that BCT may have been completely eliminated from Strawberry Creek. Immediately after the fire, efforts were focused on locating surviving BCT and relocating any that were at risk of succumbing to flash floods, siltation, and/or changes in water chemistry. BCT were discovered at three distinct locations. A total of 251 BCT were removed from the two lower locations and released in the nearby, recently restored Silver Creek. An additional 27 BCT were captured from the lowest location and released in the headwaters of Strawberry Creek above the burn perimeter. Population and habitat surveys were conducted to establish a baseline of post-fire conditions. BCT were present at only 2 of the 9 survey sites, had an average density of 114 fish/mile, and a maximum density of 268 fish/mile. This represented a 75% decrease in BCT distribution and an 85% decrease in population size. The General Aquatic Wildlife System (GAWS) for habitat surveys produced a Habitat Condition Index score of 48.6, which was a 22% decrease from the last GAWS survey taken in 2009. Habitat restoration efforts focusing on increasing ground cover, stabilizing the stream channel, and restoring riparian vegetation are being conducted. Hopefully, as the habitat recovers the BCT will as well.
Ritter, Thomas David	Salmonid Movements and Thermal Hydrodynamics at a Montane River System Confluence: Thermal Refugia in the Smith River Basin Abstract: The Smith River is a popular recreational sport fishery in western Montana, but salmonid abundances there are relatively low and limited by high summer water temperatures. Tenderfoot Creek, a major tributary of the Smith River, was the subject of a detailed multi-year study evaluating its importance to salmonids, particularly as a thermal refuge. Contrary to expectations, Tenderfoot Creek was not used as a temporary thermal refuge during periods of high water temperatures; rather, the cool outflow of Tenderfoot Creek in the Smith River was used instead. Outflow water temperatures were lower than those of the Smith River during summer; the mean difference between temperatures in the Tenderfoot Creek outflow and the Smith River outside of this plume was 2.9 °C and ranged from 0.5 °C to 6.1 °C. Detections in the cool water outflow of Tenderfoot Creek by PIT-tagged fish increased when conditions in the Smith River became stressful. Additionally, more PIT-tagged fish were detected in this plume (N = 50) than the area on the opposite bank (N = 12), which ostensibly afforded better cover. The Tenderfoot Creek outflow, as well as those of other coldwater tributaries, may be critical for salmonids when water temperatures in the main-stem river are stressful.
Rogers, Karli	Assessment of Brook Trout Passage Through Ambiguous Culvert Barriers in Pennsylvania Headwater Streams Abstract: Habitat fragmentation driven by human activity is a common threat to aquatic organisms. Road culverts in particular can isolate fish populations and reduce genetic diversity by preventing access to upstream spawning habitat. The prioritization process for removing culverts and restoring connectivity includes an assessment of passability. Culverts often receive scores that categorize them as partial barriers, known as “gray” culverts; however, detailed assessment of passability on gray culverts is lacking. To fill this research gap, we used stationary PIT-tag readers to investigate Brook Trout <i>Salvelinus fontinalis</i> passage through three gray culverts and a reference stream lacking a culvert for 16 months in Little Bear Creek, Pennsylvania. Results indicate significant differences in upstream movement rates among culvert sites. The rate of upstream passage was five times greater through the metal corrugated culvert than the reference stream. In contrast, relatively little upstream movement occurred through the two box culverts (up to 13 times less passage than the reference), indicating drastic passage differences in culverts receiving similar passability scores. Our study implies that more nuanced culvert classifications may be needed to accurately reflect fish passage.

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Roth, Curtis J.	Survival of Yellowstone Cutthroat Trout Exposed to Air During Mid-Summer Angling Events Abstract: Despite the success of catch-and-release regulations, exposing fish to air during release is a recent concern. The purpose of this study was to evaluate the effect of mid-summer air exposure on survival of Yellowstone Cutthroat Trout <i>Oncorhynchus clarkii bouvieri</i>. Yellowstone Cutthroat Trout were sampled by angling on a tributary of the South Fork Snake River, during August 2016. After capture, fish remained in the water while they were measured and tagged with T-bar anchor tags. Anglers were placed into groups of two to four people. For each angling group, the first fish captured was randomly assigned to and exposed to air for 0, 30, or 60 s. Air exposure treatments were then cycled in ascending order. In total, 328 fish were sampled (0 s, n = 110; 30 s, n = 110; 60 s, n = 108). Two weeks after angling, single-pass backpack electrofishing was used to recapture tagged fish. No difference in survival was observed among treatments (0 s, n = 75, 69%; 30 s, n = 63, 57%; 60 s n = 66, 61%), suggesting that mortality from air exposure is not a concern in catch-and-release fisheries.
Roth, Curtis J.	Let's Get Real: Air Exposure Times of Wild Trout in a Catch-and-Release Fishery Abstract: Fishing regulations are used by natural resource agencies to accomplish a variety of management objectives, including a focus on improving the quality of a fishery. Catch-and-release regulations are one of the most commonly implemented types of fishing regulations. In recent years, concerns have been raised regarding anglers exposing fish to air during catch-and-release angling. To date, there has only been one study that has explicitly focused on air exposure times of angled fish. The purpose of our study was to evaluate the length of time angled fish were exposed to air by anglers in a typical catch-and-release fishery. Anglers were observed on the South Fork Snake River from May through August 2016. Observations were conducted discreetly so as to not alter angler behavior. We recorded a number of angler characteristics including air exposure duration, fight time, approximate age of the angler, sex of the angler, how the angler accessed the river, whether the angler was guided, whether the angler took a picture, and whether the angler used a landing net. In total, we observed 316 individual anglers. We observed an average air exposure duration of 19.3 s ($SE = 0.8$), and average fight time of 40.5 s ($SE = 3.5$). Results of this study suggest that anglers typically do not expose fish to air long enough during catch-and-release angling to incur the negative effects associated with prolonged air exposure.
Ruetz III, Carl R.	Seasonal Ecology of Brown Trout in a Michigan Stream Abstract: Winter is hypothesized to be harsher than other seasons for stream fishes (i.e., reduced growth, survival, and movement) in temperate regions because of adverse conditions associated with low temperatures. To test this hypothesis, we compared movement, survival, and growth of Brown Trout <i>Salmo trutta</i> between summer and winter. We conducted capture-recapture sampling over two 7-week periods (July-September 2016; January-March 2017) in Stegman Creek, Michigan. Fish were individually marked with passive integrated transponder (PIT) tags and recaptured via weekly electrofishing surveys. Total length, mass, location within 100-m study reach (10-m increments), and tag identification were recorded for each fish. In total, we tagged 160 Brown Trout during summer and 168 during winter sampling. Tag retention (evaluated by fin clipping each PIT tagged fish) was 98.8% during summer and 97.9% during winter. In summer, Brown Trout moved an average of 6.2 m/week ($SD = 9.2$) based on recapture locations, which provide a conservative estimate of movement. Using a Cormack-Jolly-Seber model, apparent weekly survival probability increased with Brown Trout size; it was 0.75 ($SE = 0.06$) for small (80-100 mm TL), 0.86 ($SE = 0.03$) for medium (100-210 mm), and 0.93 ($SE = 0.02$) for large (>210 mm) fish during summer. The weekly recapture probability was constant among size classes (0.57, $SE = 0.03$) during summer. The length-mass relationship for Brown Trout during summer was $M = 9.6 \times 10^{-6} * L^{3.0}$, where M is wet mass (g) and L was total length (mm) at first capture. Upon completion of winter surveys, we will make seasonal comparisons to determine whether trends are consistent with the hypothesis of lower movement, survival, and growth during winter in temperate regions.
Swallow, Kyle	Combining Public Input with Science Based Management in an Arkansas Trout Fishery Abstract: Greers Ferry Tailwater (GFTW) provides a year-round trout fishery for stocked Rainbow Trout <i>Oncorhynchus mykiss</i> and is home to the only self-sustaining, wild Brown Trout <i>Salmo trutta</i> population in Arkansas. As a result, this fishery supports a diverse group of stakeholders, including

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	harvest oriented anglers, catch-and-release anglers, trophy anglers, professional guides, and trout fishing resort owners. In 2016, the Arkansas Game and Fish Commission (AGFC) revisited the GFTW Management Plan. While public input is a large component of management plan development in Arkansas, the conflicting viewpoints of the stakeholders presented several challenges. The biggest challenge was that the public perception of the fishery was often in opposition to biological and angler survey data collected by AGFC from 2007 to 2015. Many anglers expressed concerns about low catch rates and overharvest of trout, and wanted to see an increase in Rainbow Trout stocking numbers. However, data collected by AGFC indicated that both electrofishing catch rates and angler catch rates of trout were higher than previous years and harvest rates had been steadily declining since the early 1990s. Some anglers also wanted more protective regulations, while others thought that the tailwater was already being over-regulated. In an effort to make compromises that would satisfy the different user groups, AGFC recommended reduced stocking rates and more liberal harvest regulations (i.e., removal of a protective slot limit), but added three catch-and-release areas to help protect large trout. This process serves as a good example of cooperative decision-making using both biological data and public input to provide management recommendations for a controversial fishery.
Thorne, David	West Virginia Wild Trout Stream Restoration Abstract: Recent multimillion dollar stream restoration and trout habitat improvement projects could not have been completed without multiple-partner collaboration in West Virginia. State and federal agencies, nongovernmental organizations, academic researchers, and private companies are involved in major stream restoration projects to better utilize the available knowledge base to make broad-scale improvements to native Brook Trout <i>Salvelinus fontinalis</i> that would not have been possible without extensive collaboration. Large-scale stream restoration and habitat enhancement projects encompassing many entities of ownership, funding, and skillsets make project management difficult, but they can be successfully completed through multidisciplinary partnerships.
Weathers, T. Casey	Multi-geographical Metapopulation Assessment of Southern Appalachian Brook Trout Abstract: Current and historical habitat degradation combined with the introduction of nonnative salmonids have reduced the range of native Brook Trout <i>Salvelinus fontinalis</i> in the eastern United States. A critical knowledge gap exists as to how habitat fragmentation has impacted Brook Trout metapopulation dynamics among various spatial scales and distances, and whether or not such dynamics can be feasibly enhanced to facilitate dispersal within and among impaired corridors and habitats. To identify unique regionally (WV, TN, and NC) specific landscape features that affect metapopulation dynamics, we used a series of Mantel tests (full and partial) to determine the effect of isolation-by-distance and isolation-by-barriers that could be influencing pairwise population genetic differentiation ($F'ST$). In addition, we conducted multiple regression on distance matrices (MRM) analysis that incorporated $F'ST$ response to determine the roles of ecological (i.e. percent forest cover, geology, presence of nonnative salmonids, etc.) and spatial (i.e. distance among sampled populations) factors. MRM models identified regionally specific spatial patterns in genetic variation, controlled for landscape similarity, among sampled stream reaches, basins, watersheds, and regions. Our results identified landscape features impacting Brook Trout metapopulation dynamics (i.e. removal of dispersal barriers, removal of nonnative salmonids, etc.) and provide an analytical pipeline useful to connecting fragmented population segments in stream restoration projects. This information will help ecosystem managers to make progressive steps toward managing and conserving populations, enhancing dispersal through corridor connectedness, and potentially increasing genetic and phenotypic diversity and adaptive selection through increased habitat availability.
Weathers, T. Casey	Brook Trout Retain Similar Phenotypes Despite Isolation and Neutral Genetic Drift Abstract: Isolated populations of wild southern Appalachian Brook Trout <i>Salvelinus fontinalis</i> experience a multitude of high elevation, habitat specific stochastic stressors that promote genetic drift and exaggerate neutral genetic differentiation. We were interested in whether populations located in Great Smoky Mountains National Park isolated headwater stream fragments had developed uniquely identifiable phenotypes as a result of such isolation and drift. For this investigation, we used 13

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	putatively neutral microsatellite loci to examine 35 spatially isolated populations and found considerable genetic differentiation (F_{ST}: 0.02-0.94) among pairwise Brook Trout populations comparisons. However, we observed diminishing patterns of morphometric and meristic (phenotypic) differentiation at broader spatial scales. Hierarchical random effects ANOVA revealed considerable phenotypic variability within and among populations, there was no variation that distinguished fish in different basins and watersheds beyond the variation shown by populations within each basins and watersheds. Moreover, least square means analyses showed no consistent pattern of differentiation among pairwise population comparisons. Weak, yet statistically significant relationships were also observed among certain environmental variables and phenotypic characters. Inconsistent patterns of phenotypic differentiation coupled with weak associations with coarse environmental variables suggest that neutral genetic drift has not led to large phenotypic differences among populations, and local adaption or plasticity may explain some of the observed variability. Interestingly, we found no evidence for reduced phenotypic variability in populations, with very small effective population sizes (i.e., <30) or low neutral genetic diversity (allelic richness and heterozygosity).
Wegner, Justin	Impacts of Land Use on Brook Trout Thermoregulatory Effectiveness and Habitat Selection in a Michigan Coldwater Stream Abstract: Brook Trout <i>Salvelinus fontinalis</i> are native to the waters of Eastern Canada, the Appalachian Mountains, and parts of the Great Lakes Region. Today, because of widespread introductions, their distribution is primarily limited by water temperature. It is well established in the literature that cool water temperature is vitally important to many aspects of Brook Trout's life, and instances of behavioral thermoregulation during periods of warm ambient water temperature are commonly reported. We evaluated how effectively Brook Trout in a small Michigan coldwater stream maintained body temperatures within a biologically relevant range for growth, and how alterations to the riparian zone influenced Brook Trout thermoregulatory efficiency and habitat use during the summer when ambient water temperatures often exceeded their preferred range. Overall, Brook Trout body temperature conformed closely to the ambient water temperature. Brook Trout in a stream segment with an intact riparian zone maintained body temperatures within the range for growth for most of the summer; however, Brook Trout in a degraded clear-cut section had body temperatures that were warmer than mean ambient temperatures. Brook Trout in the degraded section selected habitat with greater depth than Brook Trout in the forested section that selected sites with shallower depth, cooler water temperature and greater overstory density. These findings illustrate the importance of evaluating effectiveness of thermoregulatory behavior in the context of a biologically relevant range, and accounting for the impact of proximate land use on Brook Trout thermoregulation and habitat selection when attempting to protect and restore wild trout habitat.
Wickersham, Thea	Using SNP DNA Markers for Assessing Movement and Reproductive Success of Rainbow Trout in the Buffalo River and their Contribution to the Henrys Fork Fishery Abstract: The Buffalo River is located in the Upper Snake River basin in Idaho and Wyoming and is a major contributor to the Henrys Fork River, one of the most renowned fisheries in the world. Since fish passage was greatly improved in 2006 on the Buffalo River, a significant seasonal migration between Buffalo River and the Henrys Fork has occurred. We have determined that Henrys Fork and Buffalo River Rainbow Trout <i>Oncorhynchus mykiss</i> populations exhibit sufficient genetic variation to allow the identification of unique populations. Parentage results have established evidence of Buffalo River contributing to the Henrys Fork River fishery. Our objective is to further assess the origin and diversity of Henrys Fork Rainbow Trout and resident Rainbow Trout from above the weir on the Buffalo River. These analyses should allow us to expand our baseline for better parentage assignment, and aid in understanding the life histories of the Buffalo River Rainbow Trout. If the Buffalo River is confirmed to be a major contributor to the Henrys Fork fishery, management objectives can then be formed to reflect this new information.

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Winkler, Nate	Difficulties Encountered During a Redd Survey on the Boardman River (N. Lower Michigan) Abstract: A first-ever redd survey was implemented on a northern Lower Michigan river. The river, a spring fed and baseflow driven blue ribbon trout stream is currently undergoing a multi-dam removal and channel restoration project. Salmonids present in the reaches surveyed are Brown Trout <i>Salmo trutta</i> and Brook Trout <i>Salvelinus fontinalis</i>, both of which are wild populations (no stocking). Given the inaugural nature of the survey, goals were winnowed down to the point that initially proposed methods varied from what eventually occurred both in the field and during the post-survey data analysis. Primary goals and outcomes that did mesh included provision of a spatial presence/absence analysis of spawning activity, temporal analysis of spawning activity, redd location and construction techniques, and baseline redd numbers in a newly restored section of relic channel undergoing a large wood installation project. Redd construction tended to be proximal to cover be it wood, deep water, aquatic vegetation, or along stream banks. Brown Trout created discrete, “textbook” redds in medium to large gravel while Brook Trout (potentially small Brown Trout) constructed amorphous “pots” and “smears” on various finer substrates, sometimes as uncountable numbers of pockmarks on gravelly shoals and in sediment. The primary lesson learned through this exercise is that redd surveys for Brown Trout and (especially) Brook Trout are not as simple as one might think upon reviewing the literature. Determination of what constitutes a Brook Trout “redd” especially was a difficult call to make in the field and constituted the primary confounding factor in determining overall spawning activity and determination of actual numbers of spawning Brook Trout.
Zorn, Troy	The Reintroduction of Arctic Grayling into Michigan Abstract: Though largely extirpated from the state by the early 20th century, Arctic Grayling <i>Thymallus arcticus</i> remain a cherished part of Michigan’s history and a source of fascination for many anglers. This is evidenced by several unsuccessful attempts to reintroduce the species to the state. Successful reintroductions of Arctic Grayling in Montana using remote site incubator (RSI) techniques and improved understanding of Michigan streams, have given Michigan fisheries biologists renewed hope that the species may once again be restored to Michigan waters. The Michigan Department of Natural Resources Fisheries Division and Little River Band of Ottawa Indians have formed a partnership to lead an Arctic Grayling reintroduction effort, in collaboration with Montana Fish Wildlife and Parks and other partners. The objectives of the Arctic Grayling reintroduction project are (1) To restore self-sustaining populations of Arctic Grayling to streams within the species’ historic range in Michigan; and (2) to inform future efforts to restore Arctic Grayling to formerly-occupied habitats in Michigan and elsewhere. This project is referred to as a “management experiment” due to numerous unknowns which occur in nearly every phase of the project. Since Arctic Grayling eggs are in high demand in Montana, our efforts during the first several years will focus on building a Michigan-based brood stock, working to prioritize stream reaches for reintroduction efforts, and seeking to address key research questions that will inform Arctic Grayling reintroduction efforts in Michigan and elsewhere. We provide an overview and update on this ambitious project.

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	Institution	Street	City	State	Postal Code	Phone	Email
Nate	Akey	Grand Valley State University	17018 Blackfoot Trail	Howard City	MI	49329	231-660-2038	akeyn@mail.gvsu.edu
Mike	Anderson	Montana Department of Natural Resources and Conservation	2705 Spurgin Rd. Bldg F	Missoula	MT	59804	406-542-4232	michaelanderson@mt.gov
Paul	Ayers	University of Tennessee	2506 E. J. Chapman Drive	Knoxville	TN	37996	865-974-4942	ayers@utk.edu
Juliet	Barenti	USFWS	11103 East Montgomery Drive	Spokane Valley	WA	99206	509-893-8005	juliet_barenti@fws.gov
Jeremy	Benson	University of Lethbridge	8b Modesto Rd W	Lethbridge	AB	T1K 4B8	250-417-7278	jw.benson@uleth.ca
Pat	Bigelow	National Park Service	PO Box 168	Yellowstone National Park	WY	82190	307-344-2284	Pat_Bigelow@nps.gov
Mike	Brinkley		2582 W 28th Ave	Eugene	OR	97405	408-307-1332	mbrinkle@comcast.net
Jessica	Buelow	Idaho Fish and Game	99 Hwy. 93 N.	Salmon	ID	83467	208-690-1604	jessica.buelow@idfg.idaho.gov
Sara	Bumstead	Government of Alberta	213 1 Street West	Cochrane	AB	T2T1Y9	403-932-0373	sara.bumstead@gov.ab.ca
Jason	Burckhardt	Wyoming Game and Fish Department	2820 State Highway 120	Cody	WY	82414	307-527-7125	Jason.Burckhardt@wyo.gov
Matthew	Campbell	Idaho Department of Fish and Game	1800 Trout Rd	Eagle	ID	83616	208-939-6713	matthew.campbell@idfg.idaho.gov
Bob	Carline	Retired	1244 Westerly Parkway Unit 30	State College	PA	16801	814-574-3051	m-bcarline@comcast.net
Andrew	Carlson	Michigan State University	2931 Pleasant Grove Rd	Lansing	MI	48910	651-280-7013	carls422@msu.edu
Luciano	Chiaramonte	Idaho Fish and Game	1414 E Locust Ln	Nampa	ID	83686	208-465-8404	luciano.chiaramonte@idfg.idaho.gov
Steve	Chippis	USGS South Dakota Coop Unit	1390 College Ave	Brookings	SD	57007	605-688-5467	steven.chippis@sdsate.edu
Matthew	Coombs	Alberta Environment and Parks	12501 20 Ave	Crowsnest Pass	AB	T0K 0E0	403-562-3293	matthew.coombs@gov.ab.ca
Kristopher	Crowley	Burns Paiute Tribe	100 Pasigo St	Burns	OR	97720	541-573-8083	kristopher.crowley@burnspaiute-nsn.gov
Nathan	Cullen	NPS	100 Great Basin National Park	Baker	NV	89311	775-234-7565	nathan_cullen@nps.gov
Dale	Dalrymple	AFS	3322 Cowley Way Apt 3	San Diego	CA	92117	619-275-6284	dbd@ieee.org
Dan	Dauwalter	Trout Unlimited	910 Main Street Suite 342	Boise	ID	83702	208-345-9800	ddaualter@tu.org
Jason	Detar	PA Fish and Boat Commission	595 E. Rolling Ridge Dr.	Bellefonte	PA	16823	814-359-5118	jdetar@pa.gov
Jeff	Dillon	IDFG	600 S. Walnut	Boise	ID	83707	208-287-2783	jeff.dillon@idfg.idaho.gov
Andy	Dolloff	USDA Forest Service	1710 Research Center Drive	Blacksburg	VA	24060	540-230-8220	adoloff@fs.fed.us
Sandy	Dotts	WA Dept of Fish and Wildlife	PO Box 350	Colville	WA	99114	509-684-2362	sandra.dotts@dfw.wa.gov
Emma	Easterly	University of Wisconsin-Stevens Point	2100 Main Street	Stevens Point	WI	54481	715-560-9316	emma.g.easterly@uwsp.edu
Bryan	Engelbert	Utah Division of Wildlife Resources	318 N. Vernal Ave	Vernal	UT	84078	435-219-6525	bengelbert@utah.gov
Josh	Epstein	Inter-Fluve	501 Portway Ave	Hood River	OR	97031	541-386-9003	epstein@interfluve.com
Steve	Faulkner	U.S. Geological Survey	11649 Leetown Rd	Kearneysville	WV	25430	304-724-4471	faulkners@usgs.gov
Jon	Flinders	Idaho Dept of Fish and Game	4279 Commerce Circle	Idaho Falls	ID	83401	208-535-8004	jon.flinders@idfg.idaho.gov
Seth	Fopma	SDSU	208 10th Street West	Brookings	SD	57006	641-780-8553	fopma.seth@gmail.com
James	Frazier	Molybdenite Creek	503 Summit Crossing Way	Cumming	GA	30041	858-213-3958	jrfrazier1@hotmail.com

First Name	Last Name	Institution	Street	City	State	Postal Code	Phone	Email
Mark	Fuller	USFWS	1380 S 2350 W	Vernal	UT	84078	435-828-2959	mark_h_fuller@fws.gov
Austin	Galinat	South Dakota State University	3053 Willowbend Road	Rapid City	SD	57703	605-430-4521	austin.galinat@jacks.sdstate.edu
Ashley	Gillespie	Fisheries and Oceans Canada	501 University Cres	Winnipeg	MB	r31b4	204-984-0405	ashley.gillespie@dfo-mpo.gc.ca
Bryan	Giordano	Arizona Game and Fish Department	2878 E White Mountain Blvd	Pinetop	AZ	85935	928-532-3684	bgiordano@azgfd.gov
Rob	Gipson	WY Game and Fish	PO Box 67	Jackson	WY	83001	307-690-8465	rob.gipson@wyo.gov
Christy	Graham	Arkansas Game and Fish Commission	201 East 5th Street	Mountain Home	AR	72653	870-404-0477	Christy.Graham@agfc.ar.gov
Chris	Grant	Juniata College	1700 Moore street	Huntingdon	PA	16652	814-777-1695	Grant@juniata.edu
Jennifer	Graves	Indiana University of Pennsylvania	1187 Oak St	Indiana	PA	15701	860-268-9789	gravejm10@gmail.com
Robert	Gresswell	USGS NOROCK, Emeritus	226 Silver Cloud Cir	Bozeman	MT	59715	406-522-7379	bgresswell@gmail.com
Joanna	Griffin	Wisconsin DNR	101 S. Webster St	Madison	WI	53704	608-264-8953	joanna.griffin@wisconsin.gov
Gary	Grossman	University of Georgia	Warnell School of Forestry & Natural Resources, University of Georgia	Athens	GA	30602	706-255-9082	grossman@uga.edu
Justin	Haglund	Wisconsin Department of Natural Resources	2801 Progress Rd.	Madison	WI	53716	608-221-6331	justinm.haglund@wisconsin.gov
Chuck	Hammerstad	Flycasters, Inc of San Jose, CA	780 Portswood Dr	San Jose	CA	95120	408-997-5460	chamerstad@aol.com
Paul	Harper	University of Lethbridge	944 13 Street South	Lethbridge	AB	T1J 2X2	403-393-3945	pharps@gmail.com
Brandon	Haslick	Burns Paiute Tribe	100 Pasigo St	Burns	OR	97720	541-573-8083	kristopher.crowley@burnspaiute-nsn.gov
John	Heckel	University of Idaho	875 Perimeter Drive, MS1141	Moscow	ID	83844	307-413-8759	heck5910@vandals.uidaho.edu
Bob	Heine	Ozark Outfitters	9780 E WATSON RD	SAINT LOUIS	MO	63126	314-409-2259	wildfish777@gmail.com
Gustav	Hellstrom	Department of Wildlife, Fish, and Environmental Studies, Swedish University of Agricultural Sciences	Skogsmarksgränd	Umeå		90336	-727915	gustav.hellstrom@slu.se
Fred	Henson	New York State Dept. of Environmental Conservation	23 Bloomingdale Avenue	East Greenbush	NY	12061	315-264-4838	fred.henson@dec.ny.gov
Dave	Hering	National Park Service	PO Box 7	Crater Lake	OR	97604	541-363-2910	david_hering@nps.gov
Brett	High	Idaho Department of Fish and Game	4279 Commerce Circle	Idaho Falls	ID	83401	208-525-7290	brett.high@idfg.idaho.gov
Brandon	Hoffner	Henry's Fork Foundation	PO Box 550	Ashton	ID	83420	208-270-6677	bhoffner@henrysfork.org
Jim	Johnson	Michigan Dept. Natural Resources (retired)	112 River Rd,	Ossineke	MI	49766	989-471-2913	wejay@charter.net
Clark	Johnson	Wyoming Game and Fish Department	420 N Cache St PO Box 67	Jackson	WY	83001	307-733-2383 ext224	clark.johnson@wyo.gov
Kent	Johnson	Trout Unlimited	1403 Birch Drive	Hudson	WI	54016	612-845-7258	d.kent.johnson@gmail.com
Craig	Johnson	Government of Alberta	43 Riverine Lane W	Lethbridge	AB		403-382-4346	craig.johnson@gov.ab.ca
Maureen	Kavanagh	BPA	2478 NW Savier ST Apt #2	Portland	OR	97210	503-504-8955	makavanagh@bpa.gov
David	Kazyak	USGS Leetown Science Center	2890 Poffenberger Rd	Middletown	MD	21769	304-724-4577	dkazyak@usgs.gov

First Name	Last Name	Institution	Street	City	State	Postal Code	Phone	Email
Patrick	Kennedy	Idaho Department of Fish and Game	1414 East Locust Lane	Nampa	ID	83686	208-465-8404 x271	pat.kennedy@idfg.idaho.gov
Jeremy	Kientz	South Dakota Game, Fish & Parks	4130 Adventure Trail	Rapid City	SD	57702	605-394-1967	jeremy.kientz@state.sd.us
Bruce	King	self employed	PO BOX 4258 41600 TISCHER AVE	Soldotna	AK	99669	9073942942	chinook0122@gmail.com
Matthew	Knoff	Lotek	4027 Leary Way NW	Seattle	WA	98107	425-791-1709	matt@biosonictelemetry.com
Jud	Kratzer	Vermont Department of Fish and Wildlife	374 Emerson Falls Road, Suite 4	St. Johnsbury	VT	05819	802-751-0486	jud.kratzer@vermont.gov
Kris	Kuhn	PA Fish and Boat Commission	844 Big Spring Rd.	Newville	PA	17241	717-776-3170	kkuhn@pa.gov
Matt	Kulp	National Park Service	107 Park Headquarters Road	Gatlinburg	TN	37738	865-712-0143	Matt_Kulp@nps.gov
Jamie	Laatsch	Henry's Fork Foundation	PO Box 550	Ashton	ID	83420	208 652 3567	jamie@henrysfork.org
Susanna	LaGory	Annis Water Resources Institute Grand Valley State University	2552 Dayton Dr.	Ann Arbor	MI	48108	630-536-7181	lagorys@mail.gvsu.edu
Mike	Lance	Montana State University	PO Box 173460, Montana Cooperative Fishery Research Unit	Bozeman	MT	59717	406-994-4549	michael.j.lance@gmail.com
Toby	Landeryou	Middlesex University	The Burroughs	London		NW4 4BT	+44 (0)7521 554808	m.rosenberg@mdx.ac.uk
Kathleen	Lavelle	Trout Unlimited	18 East Main Street Suite 3	Lock Haven	PA	17745	570-687-3566	klavelle@tu.org
Mike	Lien	Friends of the Teton River	P.O. Box 768	Driggs	ID	83422	208-354-3871	mike@tetonwater.org
Carol	LoSapio	Beyond Words	13198 Latigo Lane	Wellington	CO	80549	970-493-0635	carol.losapio@aphis.usda.gov
James	Losee	Washington Department of Fish and Wildlife	600 Capitol Way N	Olympia	WA	98501	360-280-1762	james.losee@dfw.wa.gov
Thomas	LoVullo	FERC	888 1st Street NE	Washington	DC	20426	202-502-8900	thomas.lovullo@ferc.gov
Andreas	Luek	University of Lethbridge	4401 University DR AWES Building	Lethbridge	AB	T1K 3M4	14033324095	andreas.luek@uleth.ca
Mark	Luttenton	Grand Valley State University	1 Campus Drive, Biology Department	Allendale	MI	49401	616-331-2675	luttentm@gvsu.edu
Gabe	Madel	Washington Department of Fish and Wildlife	600 Capitol Way N	Olympia	WA	98501	406-781-0895	james.losee@dfw.wa.gov
Dan	Magoulick	AR Coop Unit, Univ of Arkansas	Department of Biol Sci, 1 University of Arkansas	Fayetteville	AR	72701	479-575-5449	danmag@uark.edu
Edward	Maillett	U.S. Fish and Wildlife Service	5275 Leesburg Pike, MS: BMO	Falls Church	VA	22041	703-358-2322	edward_maillett@fws.gov
Benjamin	Majsterek	Quinault Indian Nation	1214 Aalis Building 'C' PO Box 189	Taholah	WA	98587	360-276-8215 ext. 7305	ben.majsterek@quinault.org
Ben	Martin	Juniata College	508 Stone Church Drive	Huntingdon	PA	16652	717-419-6794	martinbe14@juniata.edu
Joe	McGurrin	Trout Unlimited	401 Annette Ave	Stevensville	MD	21666	410-490-0570	jmcgurrin@tu.org
Jordan	Messner	Idaho Department of Fish and Game	99 Hwy 93 N	Salmon	ID	83467	208-993-1042	jordan.messner@idfg.idaho.gov
Kevin	Meyer	Idaho Fish and Game	1414 E. Locust Lane	Nampa	ID	83686	208-608-0494	kevin.meyer@idfg.idaho.gov
Diana	Miller	Wyoming Game and Fish	420 North Cache - 67	Jackson	WY	83001	307-699-3311	diana.miller@wyo.gov

First Name	Last Name	Institution	Street	City	State	Postal Code	Phone	Email
Dirk	Miller	Wyoming Game and Fish	5400 Bishop Boulevard	Cheyenne	WY	82006	307-274-2812	dirk.miller@wyo.gov
Matthew	Mitro	Wisconsin DNR	2801 Progress Rd.	Madison	WI	53716	608-221-6366	matthew.mitro@wisconsin.gov
Chuck	Mordhorst	University of Nebraska Kearney	122 W 18th St	Kearney	NE	68849	605-209-8092	mordhorstca@lopers.unk.edu
Melissa	Muradian	Henry's Fork Foundation	801 Main St	Ashton	ID	83420	208 652 3567	melissa@henrysfork.org
Lucas	Nathan	University of Connecticut	1376 Storrs Rd Unit 4087	Storrs	CT	6269	920-723-7682	lucas.nathan@uconn.edu
Kris	Nault	Arkansas Game and Fish Commission	201 East 5th Street	Mountain Home	AR	72653	501-351-8947	Kristofor.Nault@agfc.ar.gov
Bryce	Oldemeyer	Henry's Fork Foundation	512 Main St.	Ashton	ID	83420	208-652-3567	Bryce@henrysfork.org
Ben	Ortman	Henry's Fork Foundation	801 Main St	Ashton	ID	83420	706-224-2725	ben@henrysfork.org
Steve	Owens	Virginia Dept of Game & Inland Fisheries	1796 Highway Sixteen	Marion	VA	24354	276-783-4860	steve.owens@dgif.virginia.gov
Daniel	Palm	Swedish University of Agricultural Sciences	Skogsmarksgrand	UmeÅ, Sweden		90183	467-678-69176	daniel.palm@slu.se
Cavin	Park	Quinault Indian Nation	1214 Aalis Building 'C' - PO Box 189	Taholah	WA	98587	360-276-8215 ext. 7302	cavin.park@quinault.org
Bergin	Parks	Bureau of Reclamation	1150 N. Curtis Road, Suite 100	Boise	ID	83706	208-378-6202	bparks@usbr.gov
Dan	Rankin	SC DNR	311 Natural Resources Dr.	Clemson	SC	29631	864-982-2175	Rankind@dnr.sc.gov
Jacob	Rash	North Carolina Wildlife Resources Commission	645 Fish Hatchery Road	Marion	NC	28752	828-659-3324	jacob.rash@ncwildlife.org
Laura	Redfield	Idaho Fish and Game	1800 Trout Rd	Eagle	ID	83616	208-939-6713	laura.redfield@idfg.idaho.gov
Stephen	Reeser	Virginia Dept. Game and Inland Fisheries	P.O. Box 996 517 Lee Highway	Verona	VA	24482	540-248-9372	steve.reeser@dgif.virginia.gov
Travis	Rehm	South Dakota State University	424 Medary Ave South Apt. 17	Brookings	SD	57006	605-925-2063	rehm.travis@gmail.com
Jonathan	Reynolds	NPS	100 Great Basin National Park	Baker	NV	89311	775-234-7566	jonathan_reynolds@nps.gov
Thomas	Ritter	Ritter Designs	2419 Batchelor Street	Miles City	MT	59301	410-371-6866	rittercraft@gmail.com
Peter	Rodger	Fisheries and Oceans Canada	501 University Crescent	Winnipeg	MB	R3T 2N6	1-204-806-7869	peter.rodger@dfo-mpo.gc.ca
Karli	Rogers	Indiana University of Pennsylvania	315 Kingsley Street	Philadelphia	PA	19128	267-909-0851	karlimrogers@gmail.com
Curtis	Roth	University of Idaho	875 Perimeter Drive ms 1141 Suite 3	Moscow	ID	83844	1-208-792-1181	Roth7618@vandals.uidaho.edu
Garrett	Rowles	University of Nebraska Kearney	122 W 18th St	Kearney	NE	68847	605-209-5200	rowlesgw@lopers.unk.edu
Carl	Ruetz	Grand Valley State University	Annis Water Resources Institute, 740 W. Shoreline Dr.	Muskegon	MI	49441	616-331-3946	ruetzc@gvsu.edu
Shawn	Rummel	Trout Unlimited	18 E. Main St. Suite 3	Lock Haven	PA	17745	814-360-2633	srummel@tu.org
Dan	Schill	Idaho Fish and Game	600 So. Walnut St.	Boise	ID	83712	208-334-3791	dan.schill@idfg.idaho.gov
Anna	Senecal	Wyoming Game and Fish Department	PO Box 67	Jackson	WY	83001	307-733-2321	anna.senecal@wyo.gov
Predrag	Simonovic	University of Belgrade, Faculty of Biology	Studentski trg 16	Belgrade	AE	11000	+381 63 864 9340	pedja@bio.bg.ac.rs
Mark	Smith	Wyoming Game and Fish	5400 Bishop Blvd.	Cheyenne	WY	82006	307-777-4560	mark.smith2@wyo.gov
Oliva	Sparrow	University of Minnesota					651-757-5294	mcgui389@umn.edu

First Name	Last Name	Institution	Street	City	State	Postal Code	Phone	Email
Eric	Stark	Idaho Fish and Game	1414 E Locust Lane	Nampa	ID	83686	208-465-8404 ext 232	eric.stark@idfg.idaho.gov
Michael	Starr	Nevada Department of Wildlife	60 Youth Center Road	Elko	NV	89801	775-777-2332	mstarr@ndow.org
Mike	Strager	West Virginia University	4213 Ag Sci Bldg Research Way	Morgantown	WV	26506	304-276-3334	mstrager@wvu.edu
Lorie	Stroup	USDA Forest Service	1600 Pisgah Highway	Pisgah Forest	NC	28768	828-877-3350	loriestroup@fs.fed.us
Kyle	Swallow	Arkansas Game and Fish Commission	201 E 5th St	Mountain Home	AR	72653	870-424-5924	Kyle.Swallow@agfc.ar.gov
David	Sweet	Trout Unlimited	42 Sunrise Rd	Cody	WY	82414	307-899-9959	davidps992@gmail.com
Bill	Taylor	Michigan State University	1405 S. Harrison Rd., 115 Manly Miles Bldg.	East Lansing	MI	48823	517 432-5025	taylorw@msu.edu
Therese	Thompson	Western Native Trout Initiative	134 Union Boulevard Suite 665	Lakewood	CO	80228	303-236-4402	tthompson@westernnativetrout.org
David	Thorne	West Virginia Division of Natural Resources	PO Box 67	Elkins	WV	26241	304-637-0245	david.w.thorne@wv.gov
Kris	Urquhart	NDOW	380 West B Street	Fallon	NV	89406	775 423 3171 x 249	kurquhart@ndow.org
Rob	Van Kirk	Henry's Fork Foundation	P.O. Box 550	Ashton	ID	83420	208-652-3567	rob@henrysfork.org
Jennifer	Vincent	Idaho Fish and Game	1414 E. Locust Lane	Nampa	ID	83686	208-465-8404 ext. 261	jennifer.vincent@idfg.idaho.gov
David	Vetrano	Pennsylvania State University	N5920 County Road E	Bangor	WI	54614	608-486-4419	vetrano.david@gmail.com
Casey	Weathers	Grand Valley State University	321 Forest Resources Building	University Park	PA	16802	865-258-0137	tcw5095@psu.edu
Justin	Wegner	Annis Water Resources Institute Grand Valley State University	4415 Lake Michigan Dr. Apt. E511	Allendale	MI	49401	616-490-9997	wegnerju@mail.gvsu.edu
Duke	Welter	Trout Unlimited Driftless Area Restoration Effort	223 S. Rusk Avenue	Viroqua	WI	54665	715-579-7538	dwelter@tu.org
Dan	Wenk	National Park Service	PO Box 168	Yellowstone National Park	WY	82190	307-344-2009	dan_wenk@aol.com
Kevin	Whalen	USGS / CFWR Units	2327 University Way	Bozeman	MT	59715	703-269-7711	kwhalen@usgs.gov
Chad	Whaley	Grand Teton National Park	P.O. Box 170	Moose	WY	83012	307-739-3666	chad_s_whaley@nps.gov
Gary	Whelan	MI DNR Fisheries Division	P.O. Box 30446	Lansing	MI	48909	517-284-5840	whelang@michigan.gov
Shannon	White	Pennsylvania State University	413 Forest Resources Building	University Park	PA	16802	804-387-3498	slw361@psu.edu
Thea	Wickersham	Pacific States Marine Fisheries Commission	1800 Trout Rd	Eagle	ID	83616	208-939-6713	thea.wickersham@idfg.idaho.gov
Tod	Williams	National Park Service	100 Great Basin National Park	Baker	NV	89311	775-234-7560	tod_williams@nps.gov
Nate	Winkler	Conservation Resource Alliance	10850 E. Traverse Hwy, Suite 1180	Traverse City	MI	49684	231-620-4026	nate@rivercare.org
Amy	Wolfe	Trout Unlimited	18 E. Main St, Suite 3	Lock Haven	PA	17745	570-786-9562	awolfe@tu.org
Dave	Zafft	WY G & F	528 S Adams	St Larame	WY	82070	307-745-5180	david.zafft@wyo.gov
Troy	Zorn	Michigan Department of Natural Resources Fisheries Division	484 Cherry Creek Road	Marquette	MI	49855	906-249-1611 x308	zorn@michigan.gov



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