POTENTIAL METHODS TO COOL STREAMS CONTAINING APACHE TROUT IN THE WHITE MOUNTAINS OF ARIZONA AND IMPLICATIONS FOR CLIMATE CHANGE

by

Joy Elizabeth Price

A Thesis Submitted to the Faculty of the

DEPARTMENT OF NATURAL RESOURCES

In Partial Fulfillment of the Requirements
For the Degree of

MASTER OF SCIENCE
WITH A MAJOR IN NATURAL RESOURCES

In the Graduate College

THE UNIVERSITY OF ARIZONA

2013

STATEMENT BY AUTHOR

This thesis has been submitted in partial fulfillment of requirements for an advanced degree at the University of Arizona and is deposited in the University Library to be made available to borrowers under rules of the Library.

Brief quotations from this thesis are allowable without special permission, provided that accurate acknowledgement of source is made. Requests for permission for extended quotation from or reproduction of this manuscript in whole or in part may be granted by the copyright holder.

SIGNED: ________________________________

APPROVAL BY THE THESIS COMMITTEE

This thesis has been approved on the date shown below:

____ ____________________

Scott A. Bonar Date

Associate Professor of Wildlife and Fisheries Science

ACKNOWLEDGEMENTS

I would like to thank the University of Arizona for funding my research as well as the Arizona Game and Fish Department, the U.S. Forest Service, the U.S. Fish and Wildlife Service and the U.S. Geological Survey. I thank Dr. Norman Mercado Silva, Marianne Cox, Kirk Young, Stewart Jacks, Jeremy Voeltz, and Amy Unthank for their part in planning the project and acquiring the funding resources. I thank everyone at the Region 1 Arizona Game and Fish Department: especially Kelly Meyer and Mike Lopez for their support and help while conducting my field work within the Apache-Sitgreaves National Forest. I thank Carol Yde, Cindy Cowen, and Katie Hughes for all of their administrative and logistical support over the past couple of years. To those who helped me complete my field work; Lisa Trestik, Sally Petre, and Eric Baker, I couldn't have done it without you. To my committee members, Dr. William Matter and Julie Carter, your guidance and advice through this entire process have been invaluable and I have learned so much from you. Finally, I thank my advisor, Dr. Scott Bonar, for giving me this incredible opportunity. Your unwavering support through this entire process kept me going and made me a better fish biologist and for that I am forever grateful.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

The distribution of Apache trout *Oncorhynchus gilae apache*, a federally threatened species endemic to eastern Arizona, and that of other Southwestern coldwater fish species may be compressed due to increased stream temperatures associated with removal of riparian vegetation, reduced stream discharge, and higher air temperatures associated with drought and changes in climatic conditions. Knowledge of environmental conditions that best buffer streams against increases and fluctuations in water temperatures may help preserve current habitat for Apache trout. I modeled effects of select environmental variables on water temperatures, and estimated how management activities may affect stream temperatures. Using the USGS Stream Segment Temperature model (SSTEMP), I estimated how altering stream discharge, groundwater input, channel wetted width, and shade may prevent stream temperatures from exceeding thermal tolerance of Apache trout under current conditions and under a climate change scenario. Model simulations suggested increasing shade would be most effective for cooling streams, either through streamside planting or other means. Of tree/shrub species examined, individuals that provided most to least shade (mean density quality ± standard error) were ranked as follows: Douglas fir (82.40 ± 0.034) > Engelmann spruce (78.788 ± 0.032) > Ponderosa pine (76.973 ± 0.033) > Arizona alder (77.640 ± 0.034) > Bebb willow (78.131 ± 0.039) > coyote willow (67.615 ± 0.089). Feasibility of planting these species depends on availability of specific micro-site requirements. When environmental conditions will not support planting of trees, alternative shade options may be considered such as allowing thick sedge growth, shade cloth, or felled woody vegetation. Increasing groundwater input will cool streams. However, finding additional sources of groundwater for White Mountain streams may be challenging. Modeling suggested decreasing width-to-depth ratio would be most successful on streams wider than 2.0 m. Increasing stream discharge may work to lower the water temperature

a few degrees on streams with a starting discharge of more than 0.5 cms. Many complex parameters affect stream temperature, but through use of temperature models, fishery managers may evaluate how altering specific parameters, especially shade, groundwater input, stream discharge, and width-to-depth ratio could cool streams containing Apache trout or other isolated mountaintop species.

INTRODUCTION

Apache trout *Oncorhynchus gilae apache* is endemic to the headwaters of the White River, Black River and Little Colorado River drainages in eastern Arizona's White Mountains. Once abundant, the species was considered endangered under the Federal Endangered Species Preservation Act of 1967 and became federally protected in 1973 with the passage of the Endangered Species Act (Robinson et al. 2004; USFWS 2009). In 1975, the species was downlisted from endangered to threatened due to the efforts of the White Mountain Apache Tribe, Arizona Game and Fish Department (AZGFD), and the U.S. Fish and Wildlife Service (USFWS) (Rinne and Minckley 1985; Carmichael et al. 1993).

Factors Affecting Apache Trout Populations–Destruction of habitat and interactions with nonnative species have led to a decline in Apache trout populations. Forestry practices (Davies and Nelson 1994, Murphy et al. 1986), livestock grazing (Kauffman and Krueger 1984; Trimble 1994; Clary 1999), agriculture (Kraft 1972; Lenat 1984; Walser and Bart 1999), road construction (Burns 1972; Eaglin and Hubert 1993), mining (Brown et al. 1998; Rinaldi et al. 2005), and wildfires, as well as the flooding that follows fires, (Amaranthus et al. 1989; Hitt 2003; Isaak et al. 2010) can damage riparian vegetation and lead to destruction of streambank morphology and stability (Sidle and Sharma 1996; Cantrell et al. 2005). This decreases quality and quantity of spawning and rearing areas for salmonids (Platts et al. 1989; Opperman et al. 2005), alters stream discharge and temperatures (Bourque and Pomeroy 2001; Swank et al. 2001), and alters stream productivity and food supply (Hetrick et al. 1998; Robinson et al. 2004; Koetsier et al. 2007). Non-native species, such as brook trout *Salvelinus fontinalis*, brown trout *Salmo trutta*, and crayfish *Orconectes virilis* can compete with and predate on Apache trout (Carmichael et al. 1993; Rinne and Janisch 1995; Cantrell et al. 2005; USFWS 2009). Closely related species, such as rainbow trout *Oncorhynchus mykiss*, can hybridize with Apache trout compromising the genetic purity of the species (Rinne and Minckley 1985; Carmichael et al. 1993; Dowling and Childs 2002).

Apache trout have specific habitat requirements. They use instream cover in the form of woody debris, undercut streambanks, overhanging vegetation, and rocks and boulders at stream margins (Cantrell et al. 2005; USFWS 2009). They need clean coarse gravel substrates for spawning (USFWS 2009). Apache trout prey mainly on invertebrates which are usually abundant when streams are healthy (Harper 1976). Apache trout usually require water temperatures below 25 ^{o}C and sufficient shading and stream flow are usually needed to prevent lethal temperatures (USFWS 2009). Ample stream flow also helps maintain pools that are often used during periods of drought and extreme temperatures (USFWS 2009).

Temperature dynamics of Apache trout streams have not been extensively studied and management of stream temperatures through habitat modifications may allow conservation, or even expansion of Apache trout habitat (Robinson et al. 2004; Cantrell et al. 2005). Currently, fishery managers are concerned with decreases in habitat due to increases in stream temperature associated with increasing summer air temperatures and low stream discharge, further exasperated by recent drought conditions (Williams and Meka Carter 2009). Habitat fragmentation due to low flows that lead to intermittency as well as artificial barriers designed to keep nonnative salmonids from moving upstream restrict movement of Apache trout and may force them to seek thermal refuge downstream of the barrier in an area inhabited by nonnative salmonids. During May and June, before monsoon storms, air temperatures in the White Mountains are high, solar radiation is intense and the headwater stream segments above the fish barriers may approach the thermal limits of the species. In lab conditions, Apache trout can survive short increases in water temperatures up to 28 - 31 ^{0}C, (Lee and Rinne 1980; Recsetar et al. 2012) but feeding stops around 23 ^{o}C (Lee and Rinne 1980). In experiments designed to test effects of high temperature on Apache trout over extended time periods, Apache trout fry experienced approximately 40% mortality when water temperature fluctuated $\pm$ 6 ^{0}C around a 22 ^{0}C midpoint in a 30-d experiment (Recsetar 2011), corresponding to a 30-d LT_{50} of 23 ^{0}C. Because water temperature is most sensitive to changes in air temperature when stream discharge is low (Bartholow 1989), the area of stream that exceeds the LT_{50} is greatest during the 2-month low-discharge period leading up to monsoon storms.

Stream Temperature Dynamics–Stream temperature affects fish survival, growth, and reproduction (Hokanson et al. 1973; Xu et al. 2010). Heightened water temperature can lead to an increase in metabolic rates and oxygen needs of ectothermic species (Beschta 1997; Lee et al. 2003), yet warmer water holds less oxygen. Increase in the need for oxygen combined with its decreased availability may lead to an increase in stress and susceptibility to disease, and reduced ability to compete with fish species that evolved in warmer water (Beschta 1997; Cairns et al.

2005; Quigley and Hinch 2006). Fish may also exhibit reductions in growth when stream temperatures increase (Dickerson and Vinyard 1999; Marine and Cech 2004).

Stream temperatures fluctuate when energy moves in and out of the stream system (Sinokrot and Stefan 1993; Larson and Larson 1996; Beschta 1997; Sugimoto et al. 1997). The net gain or loss of energy within a stream as it flows from an upstream to a downstream point is a combination of net radiation, conduction, advection, convection, condensation, and evaporation (Brown 1983; Moore et al. 2005) (Figure 1). The majority of energy that enters the stream is through shortwave radiation from the sun (Beschta et al. 1987). Energy also enters the stream through longwave radiation from the Earth's surface, clouds, and atmosphere; through conduction between the water in the stream and the streambed (Beschta et al.1987); through advection, which results from heat exchange between surface water within the main channel and tributaries, groundwater, and hyporheic influx (Beschta et al. 1987; Constantz 1998); through convection, which is the transfer of energy across the air-water boundary (Theurer et al. 1984); and through condensation at the water surface (Hewlett and Fortson 1982). Energy leaves the stream due to longwave radiation from the surface of the water; through conduction between the stream and the streambed; through advection from incoming water; through convection across the air-water boundary; and through evaporation from the water surface (Theurer et al. 1984; Beschta et al. 1987; Hannah et al. 2004).

Energy exchanges from longwave radiation do not usually heat streams because the amount of incoming longwave radiation is usually balanced by the amount of outgoing longwave radiation over a 24-h period (Beschta et al. 1987). Energy exchanges due to bed heat conduction (Johnson 2004) and advection from hyporheic exchange (Packman and Salehin 2003) depend on sediment on the stream bed. Heat flux due to convection is a function of wind speed and air temperature (Theurer et al. 1984) and evaporation and condensation are a function of vapor pressure (Moore et al. 2005). These energy exchanges depend on climate patterns and streambed material, and cannot be easily altered by fishery biologists. Major sources of energy entering a stream that can be altered by fishery biologists are shortwave radiation and advection from incoming surface flows and groundwater (Rowe 1963; Johnson 2004).

Factors influencing amount of shortwave i.e. solar radiation reaching the stream affect stream temperature the most (Brown and Krygier 1970; Brazier and Brown 1973; Johnson 2004; Hewlett and Fortson 2007). Topography can block incoming solar radiation (Dubayah and Rich 1995), the degree depending on elevation and orientation of the sun relative to the stream (Rutherford et al. 1997). Riparian vegetation can also intercept direct solar radiation and minimize increases in stream temperature (Larson and Larson 1996). Riparian vegetation is commonly managed to reduce stream temperatures (Kauffman et al. 1997). Conifers and deciduous trees provide significant shade because of their height and extensive canopies (Beschta 1997). Sedges and rushes provide critical shading on small meadow streams (Beschta 1997). Canopy height, distance of vegetation from stream bank, and vegetation type all affect shading potential of riparian vegetation (Larson and Larson 1996; Beschta 1997).

Significant energy also enters a stream through advection from surface flows and groundwater (Ward 1985; Beschta 1987; Evans and Petts 1997; Fritz et al. 2006). Both the volume and the temperature of this inflow affect temperature in the reach. Surface water can enter a stream segment from upstream and through tributaries. Volume and temperature of incoming surface flows is affected by environmental variables upstream of the segment such as solar radiation, shade, and groundwater input. Tributaries are often smaller than the main channel and can sometimes have a warming effect (Ward 1985). Groundwater temperatures fluctuate less than temperatures of surface water, thus large inputs of groundwater can minimize daily changes in stream temperature (Evans and Petts 1997). Groundwater provides almost all surface flow in headwater streams in arid and semiarid regions during times of low precipitation (Fritz et al. 2006).

Stream discharge and width-to-depth ratio of the channel affect how quickly water temperature changes with the input of energy from solar radiation and advection from groundwater input and incoming surface flows; and like riparian vegetation, these variables can be altered by biologists (Poole and Berman 2001). Discharge is a measure of the volume of water flowing in a stream and the higher the discharge of water in a stream segment, the more energy it takes to heat it (Brown and Krygier 1970; Constantz et al. 1994; LeBlanc et al. 1997; Ruochuan and Austin 1998), and the longer it takes to cool. Streams in Modoc County, California exhibited a 0.026 ^{o}C increase in stream temperature for every cubic meter per second decrease in stream discharge (Lile et al. 2005). Furthermore, the discharge of water within a stream influences the effects of incoming surface flows and groundwater; in low discharge streams, addition of surface flows or groundwater can change the water temperature within the entire channel but in large discharge streams, surface flows and groundwater input may have less of an influence (Arrigoni et al. 2008). Since most energy exchanges occur at the air-water interface, the width-to-depth ratio of a stream strongly influences stream temperature (LeBlanc et al. 1997). A stream with a large width-to-depth ratio will have a greater surface area of stream in contact with the air than a stream with a small width-to-depth ratio, and thereby, will heat and cool faster. Furthermore, reducing stream width increases water velocity; thus decreasing the retention time of the water. The water then spends less time in contact with potential heat influxes (Johnson 2004) and therefore, will not heat up as quickly.

Anthropogenic activities often affect the main sources of energy to the stream: solar input, surface inflows, and groundwater input. Grazing and logging can lead to loss of shade (Bescheta 1997; Quinn and Wright-Stow 2008) and allow more solar radiation to enter the stream. Grazing and off-highway vehicle use can erode and degrade stream banks, leading to a channel with a greater width-to-depth ratio. This increases surface area of water subjected to direct solar radiation, longwave radiation, and convection, further contributing to increases in water temperature (Beschta 1997; Sovell et al. 2000). An increase in stream width also reduces effectiveness of shading from riparian vegetation (Bescheta 1997). Water withdrawals through irrigation diversions or groundwater pumping reduces surface inflows and groundwater input, potentially leading to increased stream temperatures (Poole and Berman 2001; Caissie 2006).

Stream Temperature Modeling–Computer programs have been developed to model how solar radiation, surface inflows, and groundwater input interact to affect stream temperature and how changes in riparian vegetation, stream discharge, groundwater input, and width-to-depth ratio alter energy entering and leaving a stream (LeBlanc et al. 1997; Rutherford et al. 1997; Chen et al. 1998; Blann et al. 2002; Whitledge et al. 2006). Several models are available to predict stream temperature under a given set of parameters (Sinokrat and Stefan 1993). The Stream Segment Temperature Model (SSTEMP) (USGS, Fort Collins, Colorado) offered several advantages for examining factors affecting stream temperatures in small headwater stream segments: (1) It was developed in 1984 and has been tested and updated numerous times (Bartholow 2002); (2) it is deterministic, meaning it is well suited for analyzing impact scenarios due to anthropogenic effects and the effects of manipulating model variables such as shade, channel width, and discharge, but it is simple enough for examining streams on which I was working – small segments, with few or no tributaries; (3) various studies have used it in field applications with accurate results (Yoshida et al. 2004; Harper-Smith and Hooper 2008; Callahan et al. 2010); (4) components of the model have been well-validated (Theurer et al. 1984; Bartholow 1989; Boyd and Kasper 2004); (5) it incorporated all of the variables, described above, that most affect temperature of Apache trout streams; (6) There were means to ground truth and calibrate the model to ensure its accuracy in the streams on which I was working.

The SSTEMP model not only allowed me to understand how changes in discharge, groundwater input, width-to-depth ratio, and shade affect stream temperatures under current climate conditions, but also under a climate change scenario. Air temperatures are predicted to increase 2 to 6 ^{0}C by 2100, depending on the amount of future greenhouse gas emissions (Karl et al. 2009). Because air temperature is related to stream temperature (Sinokrat and Stefan 1994; Johnson 2004), a rise in air temperature will further decrease available habitat for species adapted to cold water (Caissie 2006). A warming climate will have substantial effects on salmonids (Daufresne and Boet 2007; Isaak et al. 2011; Wegner et al. 2011). A 47% reduction in suitable habitat for all trout in the western United States is projected by 2080 because of climate change (Wenger et al. 2011).

Objectives–My goal was to examine how primary contributors to stream warming – solar radiation input, stream discharge, groundwater input, and width-to-depth ratio could be manipulated to cool streams in the White Mountains containing Apache trout. Specifically my objectives for this study were to:

(1) Quantify the degree that current characteristics of selected White Mountain streams; including discharge, groundwater input; width-to-depth ratio and shade; affect water temperatures.

(2) Use SSTEMP to evaluate the effect of manipulating these factors on temperatures of selected White Mountain streams containing Apache trout.

(3) Use this information to provide fishery managers a suite of options to cool White Mountain streams under varying temperature scenarios.

METHODS

Study sites–Use of SSTEMP to accurately and precisely evaluate effects of stream characteristics on stream temperatures requires selecting streams where water can flow in an uninterrupted segment, for 24-h. To measure retention time of water I used red Bright Dyes dye tablet tracers (Kingscote Chemicals, Miamisburg Ohio) and measured how long it took the water to move 100 meters and calculated how far the water would move in 24 hours. However, streams in the White Mountains are often interrupted, or are inaccessible due to reservation boundaries. Furthermore, model predictions are most accurate when inputting the most detailed measures of stream characteristics available, which require familiarity and use of precise equipment and refinement of measures to quantify stream characteristics. In addition, coverage of common riparian plant species is not uniform over the region. To get precise shade measurements, many sampling sites were needed in addition to those obtained from the final selection of streams modeled.

Therefore, my methodology consisted of three major parts: (1) refine field data collection methods and select appropriate study sites; (2) use these refined methods to quantify shade, stream discharge, width-to-depth ratio, groundwater input of selected White Mountain streams; and 3) model how these factors currently affect stream temperatures, and evaluate how altering these factors can cool streams under current climate conditions and under a climate change scenario.

In late May 2011 the Wallow wildfire, the largest in Arizona history, swept through the Apache-Sitgreaves National Forest and burned 538,049 acres, including portions of watershed I was interested in sampling. The fire was not fully contained until the monsoon season began in early July. Therefore, a necessary additional objective was to select final stream segments with a minimum of fire damage.

For part (1) of the study, during June 2010, I evaluated 11 White Mountain Apache trout streams including Bear Wallow Creek, Conklin Creek, Coyote Creek, Fish Creek, Hayground Creek, Mineral Creek, Stinky Creek, West Fork Black River, and the East Fork, West Fork, and South Fork of the Little Colorado River for possible use in the study. I tested equipment and refined procedures in an additional three lowland stream segments in April 2011: two stream segments on Cienaga Creek and one stream segment on the San Pedro River and 3 mid-elevation streams in June 2011: Canyon Creek, Christopher Creek, and Tonto Creek.

Based on the results of my initial surveys, I selected seven streams to study in detail: Conklin Creek, Hayground Creek, West Fork Black River, West Fork Little Colorado River, Canyon Creek, Christopher Creek, and Tonto Creek. Three streams did not have the necessary flow characteristics to obtain accurate model predictions: Canyon Creek, Christopher Creek, and Tonto Creek, but contained riparian vegetation that could be measured to improve accuracy and precision of shade data input in model simulations.

The streams with the necessary flow characteristics for accurate model predictions were Conklin Creek, Hayground Creek, West Fork Black River, and West Fork Little Colorado River; I selected these for model simulations. All four are located in the White Mountains of eastern

Arizona in the Apache-Sitgreaves National Forest (Figures 3 and A-1 – A-4) within the historical range of Apache trout. Mean annual air temperature for this region was 12 ^{0}C, and the warmest months were May, June, and early July. The area received a total precipitation of 568 mm in 2011, with 39% of precipitation occurring during July and August (Western Regional Climate Center n.d.). The mean value for percent possible sun during May is 78%; this is an indirect measure of cloud cover and a required input for SSTEMP.

Each stream contained one or two artificial fish barriers to prevent upstream movement of nonnative trout into Apache trout recovery areas. One segment from each stream was sampled. Each segment was located downstream of the White Mountain Apache Reservation and upstream of the artificial fish barriers. When choosing my stream segments, I avoided major tributaries or large pools when selecting stream segments, because these variables may lead to a decrease in accuracy of the SSTEMP model. I used ARC MAP (Esri, Redlands, California) and TOPO (National Geographic, Margate, Florida) to establish area of each watershed, in m^2, for each stream segment.

I sampled stream segments in mid to late May during periods of extreme low flow (Figure 2). May discharge in 2011 and 2012 was among the lowest 10% ever recorded in White Mountain streams (Figure 2). I sampled West Fork Little Colorado River from May 19-20, 2012. This segment ran through dense mixed conifer and Ponderosa pine *Pinus Ponderosa* forest (Figure A-1). Riparian vegetation was dominated by Engelmann spruce *Picea engelmannii.* The West Fork Little Colorado River was largest of the four stream segments; it was the widest and longest segment with the highest level of discharge. Area of its watershed was 32,912,260 m^2, 89% of which was forested. This stream was unaffected by the Wallow Fire. The length of the sampled reach was 5,056 m. The upstream point was placed downstream of a major incoming tributary (N 33^{0} 56’ 15.4”, W 109^{0} 32’ 19.2”) and its elevation was 2,889 m. A major tributary was defined as any tributary that changed the temperature of the main channel by more than 5% (Bartholow 1989). The downstream point was placed upstream of a series of beaver ponds (N 33^{0} 57’ 32.6”, W 109^{0} 30’ 38.5”) and its elevation was 2,784 m.

I sampled West Fork Black River segment from May 18-19, 2012. It had a watershed area of 87,201,200 m^2; 31% of which was forested. The segment on West Fork Black River flowed through an open meadow with scattered Engelmann spruce at the upstream point (Figure A-2). Effects of the Wallow Fire on the riparian vegetation within stream segment were minimal and had no effect on shading vegetation. Length of the stream segment was 2,215 m. The upstream point was just downstream of a densely vegetated canyon where the stream flowed into a meadow (N 33^{0} 53 44.0”, W 109^{0} 28’ 54.4”). Upstream elevation was 2,717 m. The downstream point was placed upstream of an incoming tributary (N 33^{0} 53’ 17.0”, W 109^{0} 28’ 19.8”). Downstream elevation was 2,690 m.

I sampled Conklin Creek from May 16-17, 2012. It was narrower and had less discharge than the West Fork Little Colorado River and the West Fork Black River. Conklin Creek had a watershed area of 76,127,300 $m^{2,}$ 21% of which was forested. It flowed through a dense riparian area consisting of Douglas fir *Pseudotsuga menziesii*, Ponderosa pine, Engelmann spruce, Gambel oak *Quercus gambelii*, Arizona alder *Alnus oblongifolia*, and coyote willow *Salix exigua* (Figure A-3). The Wallow Fire heavily affected Conklin Creek, the riparian area experienced significant damage. The stream segment was 3,245 m. The location of the upstream point was downstream of a large incoming tributary (N 33^{0} 40’ 16.4”, W 109^{0} 25’ 29.5”) and the elevation was 2,297 m. The downstream point was immediately upstream of a fish barrier (N 33^{0} 40’ 55.0”, W 109^{0} 26’ 53.7”) at an elevation of 2,208 m.

I sampled Hayground Creek from May 16-17, 2011. It was narrowest of the four streams with the least amount of discharge. It had a watershed area of 12,750,000 $m^{2,}$ 18% of which was forested. The stream segment flowed through an open meadow with only a few scattered Engelmann spruce (Figure A-4). Fire damage to riparian vegetation within the segment was minimal. The stream segment was 2,000 m. The upstream point was placed near the headwaters where the creek transitioned from shallow standing water to flowing water (N 33^{o} 50' 16.4", 109^{o} 28' 20.3"), at an elevation of 2,721 m. The downstream point was directly upstream of a road crossing that caused the stream to pool (N 33^{o} 50' 22.4", W 109^{o} 27' 27.9") at an elevation of 2,690 m.

The three streams sampled to obtain additional shading and riparian vegetation data were located in Tonto National Forest in central Arizona. I sampled a segment of Tonto Creek from June 11-17, 2011. The upstream point of the segment was placed downstream of the confluence with Horton Creek (N 34° 20' 20.4", W 111° 05' 45.7") at an elevation of 1680 m and the downstream point was placed upstream of a pumping station that caused pooling (N 34° 19' 41.2", W 111° 05' 40.0") at an elevation of 1616 m. The segment was 1713 m in length, 100% of which was vegetated with riparian vegetation close enough to provide shade to the stream. The dominant vegetation was composed of Ponderosa pine, Arizona alder, and coyote willow.

I sampled a segment of Christopher Creek from June 17-19 2011. The upstream point of the segment was placed downstream of a section of private land marked with an electric fence (N 34° 18' 47.1", W 111° 01' 27.9") at an elevation of 1758 m and the downstream point was placed upstream of a road crossing and the start of more private land (N 34° 18' 26.2", W 111° 02' 09.4") at an elevation of 1734 m. The segment was 1500 m in length, 93% of which was vegetated with riparian vegetation close enough to provide shade to the stream. The dominant vegetation was composed of Ponderosa pine, Arizona alder, and Gambel oak.

I sampled a segment of Canyon Creek from June 19-28 2011. The upstream point of the segment was placed downstream of a road crossing that caused the water to pool (N 34° 17’ 14.0”, W 110° 48’ 20.5”) at an elevation of 1993 m and the downstream point was placed upstream of the Fort Apache Indian Reservation boundary (N 34° 14’ 20.5”, W 110° 47’ 09.3”) at an elevation of 1881 m. The segment was 6494 m in length, 64% of which was vegetated with riparian vegetation close enough to provide shade to the stream. The dominant vegetation was composed of Ponderosa pine, Arizona alder, and Bebb willow.

A total of 9.7 km of stream totaling 90 transects in all three of these streams were sampled to obtain additional data for characterizing riparian vegetation. Although I sampled hydrologic, geometric and meteorological data in these three streams, the length of available segments was too short to allow for accurate modeling of stream temperatures. Therefore, I did not conduct model simulations on these three segments.

Model Description–I used the Stream Segment Temperature Model (SSTEMP, Bartholow 2000) to predict how various management activities such as planting vegetation, increasing stream discharge and groundwater input, and decreasing width-to-depth ratio, affect the energy entering and leaving a segment of stream, thus influencing the degree that water temperature changes as it flows from the upstream to the downstream point of the segment. The model uses a combination of data collected on site and default values supplied by the model, to estimate how characteristics of the stream segment affect water temperature. My use of the model consisted of two parts: (1) collecting accurate and precise on-site data to calibrate the model as closely as possible for the four selected White Mountain stream segments; (2) estimating how the four stream segments would heat under current climate conditions and under a climate change scenario, and how stream characteristics could be modified to cool streams under both of these scenarios.

Input data is divided into three categories: hydrology, geometry, and meteorology (Bartholow 2000). The upstream temperature of the segment is entered into the model and the model uses data input on the stream characteristics to predict the temperature at the lower end of the segment. The SSTEMP model is based on a 24 hour time frame and predicts *daily* average, minimum, and maximum downstream temperatures and, thus, assumes that upstream water temperature and meteorological data input to the model are *daily* averages and maximums. Methods for obtaining data for the three categories are described below; a complete list of model inputs and how these variables were obtained can be found in Appendix 2.

Model Inputs–*Hydrology inputs.*--Hydrology inputs include inflow temperature (temperature, in ^{0}C, of water entering the reach), segment inflow and outflow (discharge of water, in m^3/sec, at the beginning and end of the segment), and accretion temperature (groundwater temperature, in ^{0}C). I measured inflow temperature in the field with a HOBO pendant temperature/light data logger (Onset Computer Corporation, Cape Cod, Massachusetts). I placed a logger at the upstream end of each segment and secured it with rebar (1/2 in. x 2 ft.) in the middle of the stream channel, completely submerged and not in contact with substrate or any other heat sink or source (e.g., rocks, logs) (Oregon Plan for Salmon and Watersheds 1999). Loggers recorded water temperature every 15 min for 24 hours. I calculated the daily average water temperature at the upstream point and input the value into the model. To quantify segment inflow and outflow, I measured area of the water column (m^2), at the upstream and downstream point. I measured water velocity (m/s) at the same locations, with a USGS Pygmy Current Meter, Model 6205, and an AquaCalc Pro Open Channel Flow Computer (Rickly Hydrological Company, Columbus, Ohio). I multiplied the area of the wetted channel by the water velocity to determine inflow discharge (m^3/s). Mean annual air temperature (^oC) can be an accurate substitution for groundwater accretion temperature when actual groundwater temperature cannot be measured (Bartholow 2000). Therefore, I used mean annual air temperature from 2011 from the NOAA National Weather Service station in Greer, Arizona to represent accretion temperature.

Geometry inputs.--Geometry data for the SSTEMP model includes latitude, segment length, upstream and downstream elevation, a and b terms of width, and Manning's n (Bartholow 2000). I used an Opti-Logic 1000LH Laser Range Finder Hypsometer (Opti-Logic Corporation, Tullahoma, Tennessee) to calculate segment length and a Garmin GPSMAP 60CSx (Garmin International, Inc., Olathe, Kansas) to calculate latitude and upstream and downstream elevations. In SSTEMP, width is a function of flow in the form of: $W = aQ^b$: where W = width, in m, of stream, Q = mean discharge, in m^3/sec, of stream, and a and b are empirically derived coefficients. To calculate values a and b for each reach, I measured stream channel area (m^2) and water velocity (m/s), at transects placed perpendicular to the bank, every 100 m, starting at the upstream point of the segment and ending at the downstream point. I took the natural log of both stream width and discharge and performed a standard linear regression with discharge as the independent variable. The b term is the slope of the regression and I used the equation, $W = aQ^b$, to calculate the a term. Manning's n is a measure of the roughness of the streambed, which causes flowing water to slow due to friction (Bartholow 1989). I used pre-calculated values from

Chow 1959 for Manning's *n* based on channel characteristics such as water velocity, sediment composition, and vegetation because Manning's *n* changes little from stream to stream if they exhibit similar channel characteristics.

Meteorological inputs.--Meteorological data requirements for the model include air temperature, relative humidity, wind speed, solar radiation, possible sun, dust coefficient, ground reflectivity, ground temperature, and thermal gradient of the stream. I used a 4-channel HOBO micro station and a set of smart sensors (Onset Computer Corporation, Cape Cod, Massachusetts) to measure most of the meteorological data. I set the weather station near each stream segment, outside of the riparian vegetation. The station collected data every 15 min for one 24 hour period; the same 24 hour period that the instream data loggers recorded water temperature. I used a HOBO 12-bit temperature RH Smart Sensor to measure air temperature, in ^{0}C, and relative humidity, in %, and a HOBO Wind Speed Smart Sensor to measure wind speed, in m/s. I used a HOBO Silicon Pyranometer Smart Sensor to measure solar radiation. The measure for possible sun was obtained from a Meso West weather station in Flagstaff, Arizona. Because I measured ground-level solar radiation, I did not use dust coefficient or ground reflectivity (Bartholow 2002). Based on recommendations from the author of the model (Bartholow 2002), and because mean annual air temperature has been shown to be a good surrogate for ground temperature, I entered mean annual air temperature obtained from a weather station in Greer, Arizona (Western Regional Climate Center n.d.) into the model for ground temperature (Bartholow 2000). The thermal gradient, a unitless term, determines rate of heat lost or gained from the streambed to the water. Since small changes to this parameter do not affect downstream water temperature, I used the default value from the model of 1.65 (Bartholow 2002).

Shading inputs.--I measured shading parameters including: segment azimuth; topographic altitude; and vegetation height, crown, offset, and density. Detailed information on how I collected shade data can be found in Appendix 2. Segment azimuth is the general orientation of the stream reach with respect to due south. It determines which sides of the stream segment are called east and west. I used the compass on the Garmin GPSMAP 60CSx to estimate segment azimuth. Topographic altitude is a measure of the average line-of-sight angle to the horizon from the middle of the stream (Bartholow 1989). I measured topographic altitude and vegetation density, height, offset, and crown at each transect to obtain data every 100 m, starting at the upstream point of the segment and ending at the downstream point. These data were collected on both the east and west sides of the streambank.

Vegetation density was composed of two parts: continuity of vegetation along the stream (quantity) and percent of light filtered by leaves and trunks (quality). I estimated continuity through a simple presence/absence survey, at each transect on both sides of the stream, to quantify the percentage of the segment with riparian vegetation present. Vegetation was considered present if there was a tree that provided shade for the stream 35 m or closer to the streambank, along the line of the transect. This is the maximum distance that vegetation can effectively shade a stream (Bartholow 2000). The percentage of transects with vegetation 35 m or closer, for each side of the stream, is density quantity. If a shade tree was not 35 m or closer, then no other vegetation measurements were taken. I used an Extech Lux Meter, model 401025 (Extech Instruments, Nashua, New Hampshire) to calculate vegetation quality. At each transect, on both sides of the stream, I measured the amount of light that the shade tree blocked from the stream. The tree chosen to measure was the tree closest to the stream that provided the greatest amount of shade to the stream throughout the day. This is the same tree that I measured the remaining shade parameters. To measure density quality, I measured lux outside of the riparian area in full sunlight and then under the shade tree, at the streambank, to estimate the percentage of light filtered by the individual tree. I averaged the quality measurements of each transect, for east side and for the west side. I multiplied the quantity means by the quality means to achieve two vegetation density measurements, one for the east side and one for the west side.

I measured vegetation offset, in m, at the location of each transect, as the perpendicular distance from the wetted stream bank to the shade tree. I measured the height and crown, in m, of the shade tree. When I measured vegetation height, I included bank height as well. I measured vegetation crown as the width from tip of branch to tip of branch, at its widest point. I used the laser hypsometer to measure topographic altitude and vegetation offset, height, and crown (Opti-Logic Corporation 2004). I averaged the values of each transect on the east side, and each transect on the west side, to yield one value for each parameter for each side of the stream to put into the model.

I recorded species of shade tree at each transect and calculated relative abundance of each species along the stream segments. I calculated average values of vegetation density quality (amount of light filtered) crown, and height, for the 6 most abundant shade species.

Model Calibration–I entered all model inputs into SSTEMP and the model predicted the daily average, maximum, and minimum water temperatures at the downstream end of the segment. To assess how accurately the model predicted downstream temperature, I placed a HOBO pendant temperature/light logger at the downstream point of each segment. I used information obtained from the downstream temperature sensor to calibrate the model. Because I was interested in the daily maximum downstream temperature, I adjusted wind speed within the model until the predicted maximum downstream temperature matched the observed maximum downstream temperature. This is one of a few suggested methods of calibration (Bartholow 1989). I performed this calibration separately for each stream segment and each new wind speed was used for the remaining model simulations.

Temperature Scenarios Tested–Air temperature and associated meteorological data I used to calibrate the model was collected on site by my portable weather station. However, evaluation of factors for cooling streams would be improved using temperature data that was representative of the area's climate – not a snapshot of 1-2 years. Therefore, I obtained air temperature and associated data over a 12-y period from the MesoWest database courtesy of a local weather station in Greer, Arizona, supported by the U.S. Forest Service. I averaged daily and maximum air temperatures, relative humidity, and solar radiation from the hottest day of the year from 2001-2012 to account for annual variability. These were the temperatures and associated data that were entered into the model for the simulations. I will define this average of the 2001-2012 maximum air temperatures as "current climate conditions" to differentiate it from the climate change scenario also tested. Greer is one of the closest weather stations to streams containing Apache trout. However, it is located 244-305 m lower in elevation than some of the highest Apache trout streams. Using air temperature data from Greer ensured that I would be estimating worst-case warmest temperature scenarios for the higher elevation Apache trout streams.

Because an increase in air temperature necessarily leads to an increase in segment upstream temperature, I had to adjust this variable as well. The relationship between air temperature and stream temperature is not always linear (Mohseni and Stefan 1999). However, because my main objective was to evaluate success and feasibility of various management activities to cool streams under the same temperature scenario, I assumed the simplified relationship between stream temperature and air temperature based on 43 river and stream sites in 13 different countries that stated for every 1 ^{0}C increase in air temperature, stream temperature increased 0.7 ^{0}C (Morrill et al. 2005) would be adequate. Therefore, to estimate the increase in average upstream under each scenario; I first calculated the increase in average air temperature under each scenario from the mid-May air temperature recorded for calibration using the on-site weather station. I multiplied this increase in air temperature by 0.7 and this represented the increase in average upstream temperature for each scenario. This temperature was added to the average upstream temperature I measured in the field during mid-May. After entering this data into the model for both scenarios, I had estimates of downstream segment temperatures under each scenario without any adjustments of stream characteristics.

I evaluated factors that could best cool streams under three different temperature scenarios. The scenarios I modeled were (1) cooling streams 1 °C, when stream segment temperatures are at annual warmest temperatures (2) cooling stream segments below the Apache trout 30-d LT50, the temperature at which 50% of Apache trout die over a 30 day period, when stream segment temperatures are at their annual warmest (3) cooling streams below the Apache trout 30-d LT50 assuming average warmest air temperatures increase 6°C, as predicted under climate change scenarios. I first had to estimate initial stream conditions present under each scenario, and the associated downstream temperature. I then modified stream conditions in each scenario to determine which, if any, manipulation could bring the downstream water temperature of each segment to the desired temperature.

For all model simulations I assumed baseline discharge and vegetation remained unchanged from May values. Stream discharge from mid-May to late June varied only slightly because snow had already melted and no substantial amount of water entered the stream segments until the monsoon storms began in July (Figure 2). Baseline characteristics of the vegetation in the modeled stream segments in May were also similar to those in June. Although the amount of light that deciduous trees can block from the stream (density quality) can change from season to season, only 9.8% of my vegetation measurements on the four streams were from deciduous trees and the amount of light that conifers blocked did not change much over time.

I calculated the rate of longitudinal temperature change for each stream segment by dividing the change in daily average stream temperature from the upstream point to the downstream point by the distance from the upstream point to the downstream point and observed how changes in latitudinal temperature change among stream segments correlated the average discharge, stream width, and percent shade of each segment.

Manipulating Environmental Variables to Cool Streams–To quantify the habitat modifications needed to lower stream temperatures, I varied segment inflow and outflow, stream width, and shading characteristics, independently, within the model, to lower predicted annual maximum downstream temperatures 1 ^{0}C and below the LT_{50} for Apache trout. The 30d-LT_{50} of 22.9 ^{o}C is a measure of the *average* water temperature, over a 30 d period, at which 50% mortality of Apache trout occurs (Recsetar 2011). However, reduced growth (Brett et al 1969) and reproduction (Hokanson et al. 1973) occur before death and, therefore, I wanted to ensure that stream temperature within the segment never reached the LT_{50} during any part of the year; therefore, I used the model to lower predicted annual *maximum* downstream temperature 0.1 ^{0}C below the LT_{50} for Apache trout (to 22.8 ^{0}C). Lowering the maximum downstream temperature of each segment the same amount, 1 ^{o}C, allowed me to make comparisons across stream segments on how differences in discharge, groundwater input, stream width, and shade affect stream temperatures. I also examined how warming air temperatures, due to climate change or natural variability, may affect stream temperatures by increasing the average and maximum air temperatures 6 ^{0}C, the most liberal estimate of air temperature increase due to climate change (Karl et al. 2009), within the model, and performed the same alterations on discharge, groundwater input, stream width, and shade to lower predicted maximum downstream water temperature below the LT_{50}.

Stream discharge.--I estimated how an increase in stream discharge entering the segment affected maximum downstream temperature of each segment by adjusting the inflow and outflow discharges, the same amount, within the model. I calculated increase in discharge needed to lower water temperatures below the goal set in each temperature scenario.

Groundwater input.--Within the SSTEMP model, any accumulated discharge from the upstream point to the downstream point is from lateral inflow and groundwater input (Bartholow 2002). Because I made all of my measurements during base flows, several months after any major precipitation, and avoided all major tributaries, any increase in stream discharge from the upstream point to the downstream point was due primarily to groundwater inflow. Therefore, I increased segment outflow, while keeping segment inflow the same, to estimate the increase in groundwater, in cms, needed to reach the temperature goal under each scenario.

Stream width.--I adjusted stream width to estimate how decreasing width-to-depth ratio can lower stream temperatures. By adjusting the *b* term of the width to 0 within the model, *a* term of the width becomes the width of the stream (Bartholow 2002). I adjusted the *a* term of the width to estimate how changes in wetted channel width alone affect downstream maximum water temperature. I decreased the *a* term of the width, for each stream modeled, until the water temperature goal for each scenario was met.

Shade.--I estimated the amount of additional shade needed to meet the temperature goals under each scenario by increasing the percentage of *total* shade within the model. This parameter represents the percentage of the stream that is shaded by vegetation, topography, etc.

Vegetation.--For each of the 6 most common shade species found along streams in the White Mountains, I estimated the additional number of trees, per meter of stream segment, to achieve the required additional shade. I used vegetation species commonly found along the streams within the area to increase the likelihood that newly planted vegetation would survive. I used average values for crown, height, and density quality for each species, calculated from field measurements, to add trees one by one to the model. The majority of the deciduous tree values (80%) came from trees measured in the Tonto National Forest in June, and therefore, were fully leafed out at the time of sampling. All trees added in the simulations had an offset of zero meters. Each additional tree changed the average values of the shading parameters until the desired maximum downstream temperature was reached.

Water loss due to evapotranspiration.--When vegetation is planted near streams, plants may lower water levels through evapotranspiration (Bosch and Hewlett 1982). I assessed how evapotranspiration by newly planted vegetation would affect in-stream discharge. I used the mean crown for each vegetation species to calculate the area that the newly planted vegetation would comprise. I calculated the percentage of the watershed that would be covered by the additional vegetation under each scenario. I then estimated the decrease in stream depth; in mm per day, for each simulation, based on the average amount of water each plant species uses (Bosch and Hewlett 1982).

Alternative shading options.--When planting any of the 6 most common riparian shade species is not a possibility, alternative shade options can be utilized. I estimated the amount of stream segment that would need to be shaded in m^2, based on the increase in total shade percentage needed to lower the annual maximum downstream temperature of each stream segment under each scenario.

Statistical Analysis–I estimated the effectiveness of each method, i.e. increasing stream discharge, increasing groundwater input, decreasing width-to-depth ratio, and increasing shade, to achieve stream cooling based on the number of times each method successfully lowered stream temperature to the desired level, within the model, and performed a chi-square test with a null hypothesis that each cooling method is equally effective at lowering stream temperature. For shading data, I performed a one-way analysis of variance and Tukey-Kramer post-hoc analysis to determine if there were significant differences between how well each vegetation species was able to lower stream temperatures based on the number of trees required to reach the desired level of cooling.

RESULTS

Hayground Creek was the shortest segment from upstream point to downstream point, followed by the West Fork Black River segment, the Conklin Creek segment, and the West Fork Little Colorado River segment, which was the longest (Table 8).

Model Inputs–*Hydrology inputs.*--The Hayground Creek segment had the highest estimate of annual average upstream temperature followed by the Conklin Creek, West Fork Black River, and West Fork Little Colorado River segments (Table 1). The West Fork Little Colorado River segment had the greatest discharge, followed by the West Fork Black River segment, the Conklin Creek segment, and the Hayground Creek segment (Table 2).

Geometry inputs.--The West Fork Little Colorado River segment was widest, followed by the West Fork Black River, Conklin Creek and Hayground Creek segments. The natural log relationship between average stream discharge and average channel wetted width of the stream segment (Figure 4) was successfully used to derive the *a* and *b* terms for the width equation (Table 3).

*Meteorological inputs***.--**On-site meteorological data were obtained during the May sampling period (Table 4). Values were within the range of those typically measured by nearby official weather stations. All data could be used to calibrate the model to stream conditions.

Shading inputs.--The West Fork Little Colorado River segment had the highest density of riparian vegetation followed closely by the Conklin Creek segment. The Hayground Creek and

West Fork Black River segments had little riparian vegetation (Table 5). The Conklin Creek segment had the most diverse riparian tree and shrub community, whereas the West Fork Black River and Hayground Creek segments ran through meadows and intermittent conifers (Table 6).

Of the 6 most common shade trees and shrubs along the stream segments I sampled, Douglas fir provided the highest mean level of density quality, meaning it prevented more light from reaching the stream than any of the other common riparian species (Table 7). Ponderosa pine, however, was the largest tree with the greatest mean height and mean crown.

Model Calibration–The difference between actual maximum downstream water temperature measured in the field in mid-May and maximum downstream water temperature predicted by the model after all field data was entered was less than 1 ^{o}C for each stream segment (Table 8). The average difference among each of the stream segments between actual maximum downstream temperature and predicted maximum downstream temperature was 0.28 ^{o}C (SE 0.162). Wind speed was altered an average of 0.21 m/s (SE = 0.089) to adjust predicted temperature to match actual temperature.

Baseline Conditions under Modeled Temperature Scenarios–The Conklin Creek segment had the highest estimate of annual maximum downstream temperature, followed by the Hayground Creek, West Fork Black River, and West Fork Little Colorado River segments (Table 8). The estimate of annual maximum downstream temperature for Conklin Creek segment was 7.46 ^{o}C above the LT_{50} of Apache trout and 5.62 ^{o}C above the LT_{50} on the Hayground Creek segment. The estimates of annual maximum downstream temperature on The West Fork Black River and West Fork Little Colorado River segments did not exceed the LT_{50} of Apache trout. Under a climate change scenario, the estimates of annual maximum downstream temperature on the Conklin Creek, Hayground Creek, West Fork Black River, and West Fork Little Colorado River segments were 10.24, 8.22, 2.49, and 1.33 ^{0}C above the LT_{50} of Apache trout, respectively (Table 8).

The West Fork Little Colorado River segment exhibited the smallest rate of longitudinal temperature increase from the upstream point to the downstream point, followed by the West Fork Black River, Conklin Creek, and Hayground Creek segments (Table 9). The West Fork Little Colorado River segment had the greatest average discharge and highest percentage of total shade whereas the Hayground Creek segment had the smallest average discharge and lowest percentage of total shade (Table 9). The Conklin Creek segment had a higher percentage of shade than the West Fork Black River segment but a larger rate of temperature change; however, the West Fork Black River segment had a higher level of discharge than the Conklin Creek segment (Table 9). The West Fork Black River and West Fork Little Colorado River segments experienced an influx of groundwater, whereas the Conklin Creek and Hayground Creek segments lost discharge from the upstream point to the downstream point (Table 9).

Manipulating Environmental Variables to Cool Streams Under Each Scenario–*Stream discharge.*--The West Fork Black River segment required the greatest increase in incoming stream discharge to lower the estimate of annual maximum downstream temperature 1 ^{0}C, under current climate conditions, followed by West Fork Little Colorado River, Hayground Creek, and Conklin Creek segment (Table 10). Within the bounds of the model, it was not possible to decrease the annual maximum downstream temperature, under a climate change scenario, below the LT_{50} for Apache trout on the Hayground Creek and West Fork Black River segments by increasing incoming stream discharge.

Groundwater input.--Within the bounds of the model, I successfully lowered the estimates of annual maximum downstream water temperature of each stream segment, 1 ^{o}C and below the LT50 of Apache trout, under current climate conditions and under a climate change scenario by increasing groundwater input. The Hayground Creek and Conklin Creek segments required a smaller increase in groundwater input, in cms, than West Fork Black River and West Fork Little Colorado River segments to lower stream temperatures the same amount (Table 11). However, the Hayground Creek and Conklin Creek segments required a larger *percent* increase in groundwater input than the West Fork Black River and West Fork Little Colorado segments to lower stream temperatures the same amount (Table 11).

Stream width.--The Conklin Creek segment required the largest decrease in stream width to lower the estimate of annual maximum downstream temperature 1 ^{0}C, followed by West Fork Black River and West Fork Little Colorado River segments (Table 12). It was not possible to lower the estimate of annual maximum downstream temperature of the Hayground Creek segment even 1 ^{0}C by decreasing stream width. It was also not possible to decrease the annual maximum downstream temperature below the LT_{50} for the segments on Hayground Creek and Conklin Creek by decreasing stream width under any of the scenarios (Table 12).

Shade.--The West Fork Black River segment required the greatest increase in shade to lower the maximum annual downstream temperature 1^{o}C, followed by Hayground Creek, Conklin Creek, and West Fork Little Colorado River segments (Table 13). By increasing shade, it was possible to lower annual maximum downstream temperature of each stream segment below the Apache trout LT_{50}, under all scenarios (Table 13).

Vegetation.--Ponderosa pine provided the most shade, per tree, of the 6 most common tree/shrub species, while Coyote willow provided the least. This was based on the amount of trees, per 100 meters of stream, needed to lower the estimates of maximum downstream temperature of each stream segment 1 ^{0}C (Table 14). When tree species were compared within each of the stream segments and not across all stream segments, there was a significant difference between the ability of each species to lower annual maximum downstream temperature 1 ^{0}C (one-way blocked ANOVA with Tukey-Kramer post-hoc analysis: $F = 39.0076$, $df = 5$, $P < 0.0001$). Douglas fir, Ponderosa pine, and Engelmann spruce provided significantly more shade than Arizona alder, and Coyote willow and every vegetation species provided significantly more shade than Coyote willow (Table 15). An average of 57% (SE 4.51) more Arizona alders and Bebb willows than conifers was required to lower maximum downstream temperatures the same amount and an average of 172% (SE 13.40) more coyote willows than conifers.

To lower the estimate of annual maximum downstream temperature 1 ^{0}C, Hayground Creek required the highest number of trees, per 100 meters of stream, followed by the West Fork Black River segment, the Conklin Creek segment, and finally the West Fork Little Colorado River segment (Table 14).

By adding riparian vegetation to the model, I was able to cool the estimated maximum downstream temperature of each stream segment below the LT_{50} under current climate conditions (Table 16). However, using only coyote willow, I was unable to lower the estimated maximum downstream temperature of the Conklin Creek segment below the LT_{50} (Table 16). The estimated annual maximum downstream temperature was cooled from 30.36 to 24.38 ^{o}C, still 1.48 ^{o}C above the LT_{50}. After accounting for a 6 ^{o}C increase in air temperature, planting any species of riparian vegetation was still successful at lowering the West Fork Black River segment and the West Fork Little Colorado River segment below the LT_{50} (Table 17). On the Hayground Creek segment, it was possible to use Ponderosa pine, Douglas fir, and Engelmann spruce, independently, to lower the estimated maximum downstream temperature 0.1 ºC below the LT_{50} after a 6 ºC increase in air temperature (Table 17). However, entering the maximum number of Bebb willow into the model resulted in an estimated annual maximum downstream temperature of 22.89 ºC, which is only .01 ºC below the LT_{50} for Apache trout. Entering the maximum number of Arizona alder into the model resulted in an annual maximum downstream temperature of 22.96 ºC, 0.06 ºC above the LT_{50}, and entering the maximum number of coyote willow into the model resulted in an estimate of annual maximum downstream temperature of 24.13 ºC, 1.23 ºC above the LT_{50}. On the Conklin Creek segment, under a climate change scenario, entering the maximum number of any vegetation species into the model failed to lower the estimate of maximum downstream temperature below the LT_{50} (Table 17). Entering the maximum number of trees into the model resulted in an average decrease in the estimated annual maximum downstream temperature of 7.12 ºC, for each vegetation species. This was still 3.12 ºC, on average, above the LT_{50} of Apache trout.

Water loss due to evapotranspiration.--The amount of riparian vegetation needed to cool the estimate of maximum downstream temperature of the Hayground Creek segment 1 ºC, below the LT_{50} of Apache trout under current climate conditions, and under a climate change scenario would decrease stream depth an average of 3.5% (SE = 0.737) (Tables 18-20). The average percent decrease in stream depth for the Conklin Creek segment, the West Fork Black River segment, and the West Fork Little Colorado River segments under the same scenarios was 0.25% (SE = 0.077), 0.083% (SE = 0.02), and 0.08% (SE = 0.009), respectively. The maximum percent decrease in stream depth (10.13%) would occur on the Hayground Creek segment, under a climate change scenario, if Ponderosa pine were added to the model. Adding conifers to the stream segments would, on average, lower stream depth 52% (SE = 1.66) more than adding willows and alders.

Alternative Shading Options.--If total shade, provided by alternative methods is considered, the West Fork Little Colorado River segment would need the largest amount of additional total shade, in m^2, to reduce the estimate of annual maximum downstream temperature 1 °C, followed by West Fork Black River, Conklin Creek, and Hayground Creek segments (Table 21). To reduce the estimate of annual maximum downstream temperature below the LT_{50}, after a 6 °C increase in air temperature, Conklin Creek required the most additional shade in m^2 followed by West Fork Black River, West Fork Little Colorado River, and Hayground Creek segments (Table 21).

Success of Cooling Methods–Within the confines of the model, success of altering stream discharge, width, shade, and ground water input to cool streams in all scenarios differed (chi-square = 12.798, df = 4, P = 0.0123). Increasing total percentage of shade along the stream segment was successful for all 10 method/stream simulation combinations (Figure 5). Using riparian vegetation to achieve this level of shade was successful for 9 of 10 simulations. Increasing the amount of groundwater input within the segment was successful for all 10 simulations. Increasing the incoming discharge to the stream segment was successful for 8 simulations and decreasing stream depth was successful for only 6 simulations. Despite success within the model, choosing which option will work best for each stream segment is often dependent upon feasibility.

DISCUSSION

Maximum daily stream temperature has been found to be an important factor when determining the suitability of a stream for trout (Barton et al. 1985). Elevated stream temperatures can negatively affect salmonids (Sinokrot and Stefan 1993; Beschta 1997; Marine and Cech 2004; Cairns et al. 2005). Because of the importance of stream temperature, many studies have been conducted to understand which variables affect water temperature and how adjusting these variables can keep stream temperatures from heating to critical levels (Sugimoto et al. 1997; Blann et al. 2002; Gaffield et al. 2005; Whitledge et al. 2006). The characteristics I examined in my study, stream discharge, groundwater input, width-to-depth ratio, and shade have been found to have the greatest impact on stream temperatures.

Regression analysis was used to test if there was a significant linear relationship between average discharge of stream segment and the rate of increase in water temperature as it flowed downstream, or longitudinal temperature increase. The greater the amount of discharge, the smaller the rate of temperature increase over distance (R^2 = 0.993, F (1,3) = 294, P = 0.003) (Figure 6). Shade had a secondary effect on longitudinal temperature increase. The West Fork Black River and West Fork Little Colorado River segments had similar average levels of discharge, average stream width, and positive influxes of groundwater; however, the West Fork Black River segment ran through a meadow whereas the West Fork Little Colorado River segment ran through a Ponderosa pine and Engelmann spruce forest. Longitudinal temperature increase for West Fork Black River was 15% greater than that of the West Fork Little Colorado River. This demonstrates that small streams with low levels of discharge and meadow streams with little or no vegetation are especially sensitive to increases in stream temperature due to input of solar radiation.

Before stream restoration projects are undertaken, stopping or alleviating any anthropogenic activities in the area that led to the degradation of the stream is important (Kauffman et al. 1997). This may include limiting groundwater pumping and irrigation diversions, prohibiting off-highway vehicle use, limiting logging near riparian areas and fencing areas to prohibit grazing from both livestock and wildlife. This can help prevent further stream degradation and help ensure that restoration efforts are successful.

Stream segments with the greatest amount of current groundwater input required the smallest *percent* increase in groundwater input to cool them. Although lowering estimated maximum downstream temperature of each stream segment below the LT_{50} by increasing groundwater input was theoretically possible, the massive percent increase in groundwater input required to cool the Conklin Creek and Hayground Creek segments below the LT_{50}, under any scenario was unrealistic. However, increasing groundwater input may be feasible for cooling the West Fork Black River and West Fork of Little Colorado River segments.

The interaction between a stream and groundwater depends on the position of the stream with respect to groundwater flow, geologic characteristics of the streambed, and climatic variables (Winter 1999). Exchanges between groundwater and surface flows depend on the permeability of the hyporheic zone which depends on the hydraulic conductivity of the streambed (Brunke and Gonser 1997). The Conklin Creek and Hayground Creek segments were losing discharge from the upstream point to the downstream point indicating groundwater input was not appreciably contributing to flow in these segments. Therefore altering streambed conductivity of these segments would not be practical for increasing groundwater input. Fishery managers should concentrate efforts to increase groundwater input on stream segments that already receive some level of groundwater input, like the West Fork Black River and West Fork Little Colorado segments.

To increase the amount of groundwater input to a specific section of stream increased groundwater recharge may be attempted. Near White Mountain streams, ground water pumping and irrigation diversions may not substantially affect streamflow. Thinning upland vegetation within the watershed containing the stream segment of interest will decrease the amount of water used by the vegetation (Rowe 1963; Hibbert 1965; Bosch and Hewlett 1982) and may increase water yield available to the stream channel and will be further discussed below. Efforts should be focused on trees that do not provide shade to the stream and use a significant amount of water in order to achieve desired results, more groundwater input and cooler stream temperatures. Additional methods to increase available ground water have included artificial recharge systems and cloud seeding. Basins, furrows, and ditches can be used to pond surface waters from precipitation events over permeable surface soils that will allow infiltration into aquifers (Bouwer 2002). Cloud seeding has been used to increase precipitation during the rainy season (Bouwer 1988). This may also lead to a greater rate of groundwater recharge, especially if used in conjunction with forest thinning and artificial recharge systems.

The amount of surface water discharge entering the segment from upstream can help cool the segment, but the effectiveness of this method can vary with conditions. Under current climate conditions, it was possible to lower the modeled maximum downstream temperature of each stream segment below the LT_{50}. However, under a climate change scenario only the Conklin Creek and West Fork Black River segment temperatures could be lowered below the LT50. Even though lowering the maximum downstream temperature of each segment below the

LT_{50}, was theoretically possible within the model, the additional increase in discharge to the segment was too unrealistic to be a feasible option under any scenario.

The volume of water moving into the segment is affected by a variety of variables upstream of the segment: amount of precipitation, amount of vegetation, streambed composition, and groundwater and surface flow inputs (Swift and Swank 1981; Roulet 1990; Storck et al. 1998). Because the amount of water within headwater streams is heavily dependent on groundwater input during low flow periods (Roulet 1990), any action that increases groundwater recharge upstream of the segment will increase the water flow into the segment. This includes limiting ground water pumping and irrigation diversions and removing woodland vegetation upstream. A review of 39 studies reported that water yield could increase up to 4.5 mm per year with each percent reduction in forest cover; however, most catchments produced less than half this amount (Hibbert 1965). The amount of increase in streamflow with the removal of vegetation depends on climate, vegetation, soil, and capability of watershed for yielding increases (Rowe 1963). Vegetation removal treatments should be focused on areas with water supply adequate to exceed evapotranspiration loses after treatment, areas where the zone of saturation is within reach of the vegetation using the most water, and areas where the soils above the water table are of sufficient extent and depth to permit reduction in evapotranspiration if the vegetation is removed (Rowe 1963). This method would be better suited to canyon streams with a high density of vegetation such as Conklin Creek and West Fork Little Colorado River. On the West Fork Black River segment, upland vegetation could be thinned upstream of the meadow. This would increase the amount of streamflow entering the warmer meadow reach.

Although vegetation removal has resulted in increased water yield in many studies, some concerns about the practice exist. Ellison et al. (2012) suggest that thinning trees does not always lead to increased water yield, and cite evidence that presence of forests increases the intensity of the hydrologic cycle, thus providing more precipitation to an area. Ultimately they argue that water yield from tree thinning may be a function of catchment scale. In large catchments, sizable tracts of trees can significantly influence the hydrologic cycle, while in smaller catchments, thinning may have positive results on water yield. Furthermore, increased water yield through vegetation removal can be short-lived. Unless continued removal occurs, regrowth will rapidly take up the excess water yield. Whatever the thinning strategy, any cutting efforts should be focused on trees that do not shade the stream and on those that use a significant amount of water to achieve desired results.

Narrowing and deepening the stream (i.e., decreasing the width-to-depth ratio) can sometimes have a greater effect on stream temperature than increasing riparian vegetation (Blann et al. 2002). My findings suggest that wide stream channels with the greatest discharge require the smallest decreases in width to lower downstream water temperatures. Model simulations suggest that narrowing the stream channel to cool stream temperatures below the LT_{50}; both under current climate conditions and after a 6 °C increase in air temperature; would only be feasible in the West Fork Black River and West Fork Little Colorado River segments.

A smaller width-to-depth ratio means that less surface area of the stream is in direct contact with the air, resulting in less heat entering the stream through solar radiation and convection. To narrow a stream, managers can implement passive methods to allow the stream to narrow on its own such as: eliminating sources of bank erosion; fencing to prevent streamside grazing by cattle and wildlife, thus encouraging growth of dense vegetation on exposed stream banks to trap sediments; and reducing sources of sedimentation that lead to a shallower streambed. Managers can also actively decrease width-to-depth ratio by using large boulders, logs, and root wads to stabilize banks and narrow the channel (Rosgen 1997). Using natural materials can off-set some of the adverse aesthetic and biological impacts associated with streambank stabilization (Rosgen 1997). Planting grasses and sedges along the streambanks may be another option for decreasing stream width. Grasses along streambanks have been shown to store sediments along the channel (Trimble 1997). This not only promotes encroachment of the stream, making it narrower, but decreases downstream sediment yields, which can also negatively affects salmonids (Wu 2000; Suttle et al. 2004) If a stream narrows then water velocity will increase, thus decreasing the retention time of the water. This means the water spends less time in contact with potential heat influxes (Johnson 2004) and will not heat up as quickly over a given length of stream. In extreme cases, where simpler methods fail, entire channel reconstruction may be an option (Rosgen 1985).

Shade is one of the few variables affecting stream temperature that can feasibly be altered by fishery managers (Guoyuan et al. 2012). Its influence on stream temperature depends on stream location; groundwater input; and width, composition, and density of the riparian buffer strip (Osborne and Kovacic 1993). Riparian vegetation can increase the distance it takes water to heat from groundwater temperature to the mean daily air temperature by 13-34% in spring-fed streams during summer (Whitledge et al. 2006). A previous study using SSTEMP found that increasing riparian canopy cover to 100% decreased downstream maximum temperature by ~ 10 ^{0}C, while increasing groundwater input by the maximum amount only decreased maximum downstream water temperature ~ 2 ^{0}C (Harper-Smith and Hooper 2008). Riparian vegetation is an important means for cooling streams, often surpassing other methods.

My model simulations demonstrated that streams with a higher starting percentage of total shade required less additional shade to lower annual maximum downstream temperature 1 ^{o}

C. Increasing the amount of shade was successful at lowering the annual maximum downstream temperature of all stream segments below the LT_{50} of Apache trout, under current climate conditions and after a 6 ^{o}C increase in air temperature. Possible methods of providing shade include riparian vegetation and artificial shade structures.

Model results suggest adding riparian vegetation to the stream segments would lower annual maximum downstream temperatures below the LT_{50} of Apache trout under current climate conditions and after a 6 ^{o}C increase in air temperature in every segment except Conklin Creek after a 6 ^{o}C increase in air temperature. This is likely because the Conklin Creek segment, under a climate change scenario, would require the largest decrease in stream temperature, 10.34 ^{o}C, to lower the estimate of annual maximum downstream temperature 0.1 ^{o}C below the Apache trout LT_{50}.

I measured the shade characteristics of *individual* trees, which on average prevented 76.9% of sunlight from reaching the streams (density quality). Planting vegetation in layers that might block even more light from reaching the stream, could lower stream temperatures even more, perhaps even below the Apache trout LT_{50} in Conklin Creek under a climate change scenario. To successfully lower the annual maximum downstream temperature below the LT_{50} under a climate change scenario in Conklin Creek, conifers or either of the large deciduous trees would need to block an average of 96% of light from reaching the stream. Coyote willow would need to block 98.6% of light from reaching the stream. Feasibility of planting additional trees to achieve this high degree of shading depends on how much vegetation the riparian area can support, especially during times of low flow.

Conifers provide the greatest amount of shade per tree and, therefore, would be ideal for stream shading. However, conifers take 40 years, on average, to reach the height I measured in the field and used in the model (USDA NRCS National Plant Data Team). Alders (Featherstone 2012) and willows (Nellessen n.d.) require only 10 years, on average, to reach the height I measured and used in the model. Therefore, desired level of shading is achieved four times quicker when planting willows and alders than when planting conifers.

The specific micro-site requirements of the different species must be considered when choosing riparian trees to plant. The type of riparian vegetation that an area can support depends on regional climate, stream gradient, elevation, soil, aspect, topography, water quantity and quality, type of stream bottom, and plant community (Oakley et al. 1985; Prentice et al. 1992; Richardson et al. 2007). For a specific species to become established, they need correct soil moisture levels, light conditions, temperature fluctuations, and other environmental conditions (e.g. fire). The species composition of a riparian community is governed by the conditions available (Richardson et al. 2007). For example, willows commonly occur on point-bars of newly deposited, coarsely textured, well aerated substrates and occasional high flows are needed to create these conditions (Kauffman et al. 1997). Availability of water from the water table and depth to groundwater limits the number and type of species to plant within the riparian zone, especially in meadow regions (Steed and DeWald 2003). Prolonged drought or over-pumping of groundwater can lead to mortality of riparian trees (Richardson at al. 2007). The headwater streams of the White Mountains are currently exhibiting drought conditions characterized by smaller levels of discharge, compared to non-drought conditions (Figure 2). Therefore, because the deciduous trees use less water than the conifers; it may be more feasible to plant deciduous trees than conifers. Furthermore, because many of the streams currently run through meadows, conifer growth may not be supported in those areas (Kauffman et al. 1997).

Sedges were not a common source of shade within the stream segments I studied; however, planting sedges may be another option to cool the meadow stream segments on Hayground Creek and West Fork Black River. Sedges can be an important source of shade, especially in meadow regions that cannot support the growth of large trees due to fine-textured soils and channel substrates, shallow water tables, and anaerobic soils (Bescheta 1997; Kauffman et al. 1997; Steed and DeWald 2003). The amount of shade that the sedges can produce depends on the maximum height that the sedges can reach and the width of the streambank. Further characterization and modeling of the shade provided by sedges in grazed and ungrazed areas would be beneficial.

Shade can also be provided by artificial techniques. Although use of artificial shade may not be a desirable alternative, it could be useful in areas that do not possess specific micro-site requirements to support riparian vegetation. Artificial shade provides immediate shading, it does not require water, and it may be effective temporarily when a forest fire has completely removed existing riparian shade vegetation and rendered soil too damaged to support new growth or plantings. This may be a better option when only small areas of the stream need to be shaded or used in conjunction with riparian vegetation.

Shade cloth and felled vegetation are common forms of artificial shade that can prevent stream warming (Kiffney et al. 2004; Matney 2004; Gothreaux and Green 2012). Before shading, the temperature of a stream in Rockingham County, Virginia increased 0.6 ^{o}C in 550 m and after covering the 550 m segment with shade cloth that reduced the amount of incoming solar radiation by 79%, the stream cooled 1.2 ^{o}C (Fink 2008). Shade cloth can be deployed in many ways, over the stream, or on a streambank angled over the stream on the side which receives the most direct solar radiation throughout the day. This way, solar radiation is blocked from the stream, but some light can reach the stream for important biological processes. Cut juniper placed over a small meadow stream in southeast Oregon decreased the stream temperature by 2 ^{o}C and decreased the influence of air temperature on stream temperature fluctuations (Matney 2004).

Planting permanent riparian vegetation provides benefits beyond stream cooling, benefits which are unavailable when increasing stream discharge, decreasing stream channel width, or installing artificial shade. A riparian buffer strip protects water quality by preventing sediment, nitrogen, phosphorus and pollutants such as pesticides from entering the stream (North Carolina State University n.d.). Vegetation provides energy and beneficial nutrients for stream organisms, especially in high mountain headwater streams where up to 99% of energy input comes from woody debris and leaf litter (North Carolina State University n.d.). Riparian vegetation provides food and cover for terrestrial wildlife. Vegetation also slows floodwaters and aids in stream bank stability. (North Carolina State University n.d.) Improving aquatic environments, which can enhance native sport fish abundance, increases the values of the stream and this increase in economic value can often outweigh the cost of re-vegetating the stream (Theurer et al. 1985). When factoring in additional benefits of planting riparian vegetation, this option often becomes more cost efficient than altering the channel’s width or using temporary artificial shade.

In addition to rising stream temperatures, imperiled fishes are threatened by predation and competition from non-native species (Robinson et al. 2004; Cantrell et al. 2005). Cooling the temperature of a stream to optimize it for a particular native species may limit growth of a non-native competing species, if the species of concern has an optimal growth temperature higher than the endemic species (Rahel et al. 2008). In the West Fork of the Black River, Apache trout are threatened by non-native virile crayfish *Orconectes virilis* (Inman et al. 1998; Childs 1999). *Orconectes virilis* prefer stream temperatures from 22-26 ^{o}C (Peck 1985). Crayfish may compete for food with Apache trout by impacting benthic macroinvertebrates (Fernandez and Rosen 1996; Carpenter and McIvor 1999). Therefore, keeping stream temperatures below the preferred temperature of crayfish may aid in Apache trout survival.

I did not study effects of stream substrate type on water temperature because this variable is not easily altered by fishery managers. However, stream substrate type can affect stream temperature and may be important in prioritizing where to make alterations to the stream. A bedrock stream reach exhibited higher temperature fluctuations than an alluvial reach (Johnson 2004). Smooth bedrock surfaces reflect solar radiation back into a shallow stream. However, an alluvial layer may exhibit extensive hyporheic exchange, leading to dampened daily stream temperature fluctuations. This hyporheic exchange is uncommon in bedrock lined channels (Johnson 2004). Substrate type can affect water velocity, which affects stream temperature. The quicker water moves through an area, the shorter the hydraulic retention time and the shorter the duration of contact between the water and the surrounding influences. Water moves faster over bedrock than it does over gravel or sand (Johnson 2004).

Pools can also influence the amount of available thermal habitat available for salmonids. When pools are large enough to stratify by temperature, they have provided thermal refuge for various salmonid species (Nielsen et al. 1994; Elliott 2000; Tate et al. 2007). For a pool to stratify there must be a source of cold water; this is usually from tributaries or the streambed (hyporheic or groundwater input) (Nielsen et al. 1994). If fishery managers wish to create stratified pools, than they should ensure a source of cold water is present, even during the warmest time of the year. Proximity of a pool to a source of cold water can be more important than size of pool (Matthews and Berg 1997). In order for a pool to remain stratified, mixing has to be stopped or weakened (Nielsen et al. 1994). If the velocity of the incoming cold water is not large enough to prevent mixing, than large woody debris may be placed in the stream to create an effective barrier to mixing (Nielsen et al. 1994). If a pool is deep enough, it may remain stratified if mixing is weak. One study found that pools deeper than 3 meters stratified when surface flow decreased to 1 m^3/s and the temperature at the bottom of the pools averaged 3.5 ^{o}C cooler than temperature at the surface (Nielsen et al. 1994). When streams are allowed to pool, cooler water at the bottom of the pools can enter the streambed through hyporheic flow. This cooler water can later reenter the stream at shallow riffle sites where stream temperatures may be closer to air temperature. This can be an important source of cooling during the warmer summer months (Poole and Berman 2001). Therefore, increasing the amount of pools within a stream segment may also increase the cooling effects of hyporheic flow. The downside to creating pools would be a slowing of flow, which may cause water at the surface to heat up faster. This is further motive for making sure the pools are deep enough to stratify and provide refuge for trout at the bottom from warmer surface waters.

Stream temperature is just one component that makes a stream environment suitable or unsuitable for Apache trout. Other factors have proven to be important in predicting trout abundance such as number and volume of pools, current velocity, in-stream cover, food abundance, and water clarity (Lewis 1969; Wilzbach 1985; Mesick 1988; Cantrell et al. 2005; Mellina and Hinch 2009). Apache trout have been shown to select wider and deeper pools with slower velocities, more percent eddy flows, lower width-to-depth ratios, more percent boulder and undercut bank cover, and less in-stream vegetation cover (Robinson et al 2004; Cantrell et al. 2005). In-stream cover provides security for trout while faster current velocity brings more food through organism drift, allowing a trout to use less space to obtain the required amount of food (Lewis 1969). Apache trout and brown trout preferred areas with cover and adults greater than 14 cm failed to emigrate even when food was withheld for 73 days (Mesick 1988). When stream temperatures peaked during the summer, food abundance became more important than in-stream cover in predicting the abundance of adult cutthroat trout (Wilzbach 1985). Focus on habitat requirements of Apache trout, beyond their thermal needs, optimizes areas for Apache trout growth, reproduction, and survival.

There were potential sources of error within my research. I estimated *annual* maximum upstream and downstream temperatures for each segment based on annual maximum air temperatures averaged over a 12-year period. Also, I did not measure groundwater or ground temperature in the field, but instead, used mean annual air temperature to represent these values. Mean and maximum annual air temperature have been shown to be accurate surrogates for these parameters, yet measuring them in the field would have led to more exact values. However, I wished to reach general conclusions about the area, requiring me to characterize water temperatures over a long time period to minimize annual variability. The 12-year data set of temperatures could only be provided by a nearby weather station. Streamflow in these segments, while low, had not yet reached its absolute summer minimum (Figure 2). However, the difference was slight, and using streamflow from mid-May did not affect the success of the cooling methods in relation to each other, because the same starting streamflow was used for each simulation.

Certain assumptions about the stream segment and data entered into the model are required by SSTEMP. The model assumes that water within the stream is thoroughly mixed at all times, with no vertical temperature gradient. It assumes that lateral inflow and outflow is uniformly apportioned throughout the segment, meaning there are no major tributaries. SSTEMP also assumes that the stream takes 24 hours to flow from the upstream point to the downstream point of the segment. I selected Apache trout streams to study and identified segment placement within the streams to comply with these assumptions. To avoid large tributaries and pools on the larger streams, West Fork Black River and West Fork Little Colorado River, segments were too short to adhere to the 24 hour rule. However, the Conklin Creek and Hayground Creek segments were approximately long enough not to violate the 24 hour assumption. Small violations of the assumptions within the model may have been a source of error; however, in general I found model predictions for all stream segments were quite accurate. The maximum downstream water temperature that SSTEMP predicted was on average only 0.28 ^{0}C (SE 0.162) off from the measured water temperature. The HOBO pendant temperature light loggers I used have an accuracy of $\pm$ 0.53°C from 0° to 50°C.

Models help predict effects of management actions, but are not infallible. As in all models, not all variables affecting temperature could be included in SSTEMP. The model cannot predict the cumulative effects of planting vegetation along the stream. I calculated the decrease in stream depth likely to occur after planting additional riparian vegetation. However, altering riparian vegetation can also affect channel width, air temperature, relative humidity, and wind speed (Bartholow 1989). Changes in these variables can have unknown effects on the stream temperature. Furthermore, increasing the level of discharge within the stream may widen the channel, leading to a different maximum downstream water temperature than predicted.

Problems may also arise when trying to compare results to other streams. Factors unique to a stream, including climate, terrain, species of vegetation, and soil type can lead to error when trying to predict the effectiveness of a vegetated buffer strip for altering stream temperatures (Barton et al. 1985). I measured shading parameters on site within the stream channel, because riparian vegetation can exhibit complex and variable shading patterns depending on the surrounding vegetation (Guoyuan et al. 2012). Despite possible sources of error and limitations, the 30-year history of this model and its use in numerous other field applications with accurate results indicates that it is useful for understanding how different stream cooling methods relate to one another under varying stream conditions.

SUMMARY OF MANAGEMENT OPTIONS

- Stream temperatures are most affected by incoming solar radiation. Alterations in riparian vegetation affect how much solar radiation reaches the stream, and changes in groundwater input, channel width-to-depth ratio, and in-stream discharge affect how quickly the water heats up with incoming energy.
- Before altering environmental variables, anthropogenic sources of damage should be stopped or alleviated. Possible actions to reduce or stop further damage to streams may include:
    - Fencing riparian vegetation from wildlife and livestock
    - Limiting groundwater pumping and irrigation diversions
    - Limiting logging near streams
    - Avoiding or reducing road impacts to streams
    - Controlling or limiting off-road vehicle use on streambeds and banks

- Model simulations suggested shading solar radiation was most successful at cooling stream temperatures below the Apache trout LT_{50}, followed by increasing groundwater input, increasing stream discharge, and decreasing width-to-depth ratio.
- When feasibility of methods is considered, decreasing the width-to-depth ratio becomes more effective than increasing groundwater input or stream discharge to lower stream temperatures below the Apache trout LT50.
- Planting native riparian vegetation is ideal, provides added benefits such as bank stability and prevents sediments and pollutants from entering the stream.
  - Of the 6 most common shade producing trees along the streams I sampled, Ponderosa pine provided the most shade per tree followed by Douglas fir, Engelmann spruce, Bebb willow, Arizona alder, and coyote willow.
  - Planting conifers would be ideal based solely the amount of shade they provide. However, this option is not always feasible, especially in meadow reaches, and where rapid results are needed. Regional climate, soil composition, water quality and quantity, type of streambed, light conditions, and temperature fluctuations determine which trees can be successfully established. Many references provide information on which species to plant in specific areas (Oakley et al. 1985; Prentice et al. 1992; Kauffman et al. 1997; Steed and DeWald 2003; Richardson et al. 2007).
  - Planting sedges or tall grasses, or encouraging natural growth, may be an option to shade meadow reaches that do not have the soil requirements to support trees. Further research is needed to quantify the amount of shade provided by these species.

- When planting vegetation is unfeasible because specific micro-site requirements for plants are unavailable alternative shade options may be implemented. Common forms are shade cloth and felled vegetation.
    - Felled vegetation is more natural and may be better suited for most areas where aesthetics are important.
    - Because the unaesthetic nature of shade cloth likely precludes its use on the Apache trout streams I studied, it may be an option on other streams that cannot support live vegetation and or need only a small amount of additional shade.
- If shading the stream is unfeasible, than decreasing width-to-depth ratio is the next best option. This will be more successful on streams wider than 2.0 m with discharge of more than 0.5 m^3/s. Possible methods to decrease width-to-depth ratio include:

  *Passive methods that allow for a natural decrease in width-to-depth ratio:

    - Eliminate sources of bank erosion.
    - Fence stream banks to prevent grazing by cattle and wildlife. Encourage natural growth of dense vegetation along stream banks, especially exposed banks, to trap sediment and rebuild damaged banks.
    - Reduce sources of disturbance within the watershed that lead to sediments entering the stream and decreasing stream depth.

  *Active methods to reduce width-to-depth ratio:

    - Use large boulders, logs, and root wads to stabilize banks and decrease the width-to-depth ratio of the channel.
    - Plant sedges or tall grasses that can narrow the stream channel.

  - In extreme cases, channel reconstruction projects may be implemented.

- Increasing groundwater input and stream discharge are not as highly recommended as blocking solar radiation or decreasing the width-to-depth ratio due to their ineffectiveness in many circumstances. Furthermore, the groundwater resources can be complex and out of the control of many stream managers. I found increasing groundwater input more successful on streams that already had groundwater input and increasing stream discharge more effective on the large stream segments with current discharge greater than 0.5 cms. Options to increase groundwater input and stream discharge include:
  - Reducing uptake of available water by plants by removing upland vegetation. Removing conifers in upland – not riparian areas - would be most effective since they use the most water and do not provide shade to the stream. This would increase groundwater input and stream discharge by increasing runoff to the stream and groundwater recharge.
  - If applicable, reducing or stopping stream diversions or groundwater pumping upstream of the stream segment.
  - Implementing artificial recharge systems to increase the amount of groundwater recharge in the area from precipitation events.
  - Increasing precipitation through cloud seeding.

REFERENCES

Amaranthus, M., H. Jubas, and D. Arthur. 1989. Stream shading, summer streamflow and maximum water temperature following intense wildfire in headwater streams. USDA Forest Serive General Technical Report 75-79.

Arrigoni, A. S., G. C. Poole, L. A. Mertes, S. J. O'Daniel, W. W. Woessner, and S. A. Thomas. 2008. Buffered, lagged or cooled? Disentangling hyporheic influences on temperature cycles in stream channels. Water Resources Research 44: np.

Bartholow, J. M. 1989. Stream temperature investigations: Field and analytic methods. United States Fish and Wildlife Service Biological Report 89, Fort Collins, Colorado.

Bartholow, J. M. 2000. The stream segment and stream temperature models: A self-study course. United States Geological Survey, Fort Collins, Colorado.

Bartholow, J. M. 2002. SSTEMP for Windows: The Stream Segment Temperature Model (Version 2.0). United States Geological Survey computer model and documentation. Available: http://www.fort.usgs.gov/

Barton, D. R., W. D. Taylor, and R. M. Biette. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. North American Journal of Fisheries Management 5:364-378.

Beschta, R. L., R. E. Bilby, G. W. Brown, L. B. Holtby, and T. D. Hofstra. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. Streamside Management: Forestry and Fishery Interactions 57:191-232.

Beschta, R. L. 1997. Riparian shade and stream temperature: an alternative perspective. Rangeland 19:25-28.

Blann, K. J., J. F. Nerbonne, and B. Vondracek. 2002. Relationship of riparian buffer type to warm water temperature in the driftless area ecoregion of Minnesota. North American Journal of Fisheries Management 22:441-451.

Bosch, J. M. and J. D. Hewlett, 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology 55:3-23.

Bourque, C. P. A. and J. H. Pomeroy. 2001. Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: an inter-catchment, multiple-year comparison. Hydrology and Earth System Sciences 54:599-613.

Bouwer, H. 1988. Water conservation. Agricultural Water Management 14:233-241.

Bouwer, H. 2002. Artificial recharge of groundwater: hydrogeology and engineering. Hydrogeology Journal 10:121-142.

Boyd, M. and B. Kasper. 2004. Analytical methods for dynamic open channel heat and mass transfer methodology for the heat source model. Version 7.0.

Brazier, J. R. and G. W. Brown. 1973. Buffer strips for stream temperature control. Research Paper 15. Oregon State University Forest Research Laboratory, Corvalis, Oregon.

Brett, J. R., J. E. Shelbourn, and C. T. Shoop. 1969. Growth rate and body composition of fingerling sockeye salmon, *Oncorhynchus nerka*, in relation to temperature and ration size Journal of the Fisheries Research Board of Canada 26:2363-2394.

Brown, G. W. and J. T. Krygier. 1970. Effects of clear-cutting on stream temperature. Water Resources Research 6:1133-1139.

Brown, G. W. 1983. Forest and Water Quality. Oregon State University Bookstore, Inc., Corvallis, Oregon.

Brown, A. V., M. M. Lyttle, and K. B. Brown. 1998. Impact of gravel mining on gravel bed streams. Transactions of the American Fisheries Society 127:979-994.

Brunke, M. and T. Gonser. 1997. The ecological significance of exchange processes between rivers and groundwater. Freshwater Biology 37:1-33.

Burns, J. W. 1972. Some effects of logging and associated road construction on northern California streams. Transactions of the American Fisheries Society 101:1-17.

Cairns, M. A., J. L. Ebersole, J. P. Baker, P. J. Wigington, H. R. Lavigne, and S. M. Davis. 2005. Influence of summer stream temperatures on black spot infestation of juvenile coho salmon in the Oregon Coast Range. Transactions of the American Fisheries Society 134:1471-1479.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51:1389-1406.

Callahan, M. K., J. C. Bellino, and M. C. Rains. 2010. Trends and controls on summer surface-water temperatures in salmonid-bearing headwater streams in two common geomorphic settings, Kenai Peninsula, Alaska. AGU Fall Meeting Abstracts 1:1083.

Cantrell, C. J., A. T. Robinson, and L. D. Avenetti. 2005. Habitat selection by apache trout in six east-central Arizona streams. Transactions of the American Fisheries Society 134:1382–1388.

Carmichael, G. C., J. N. Hanson, M. E. Schmidt, and D. C. Morizot. 1993. Introgression among Apache, cutthroat, and rainbow trout in Arizona. Transactions of the American Fisheries Society 122:121-130.

Carpenter, J. and C. McIvor. 1999. Effects of an introduced crayfish on native Arizona fishes. Final Report to Arizona Game and Fish Department, Heritage Program IIAPM Project No. I95034. University of Arizona, Tucson.

Chen, Y. D., R. F. Carsel, S. C. McCutcheon, and W. L. Nutter. 1998. Stream temperature simulation of forested riparian areas: I. Watershed-scale model development. Journal of Environmental Engineering 124:304-315.

Childs. M. 1999. Literature review and field and laboratory studies of Northern crayfish, *Orconectes virilis*, in Arizona. Heritage IIAPM Project No. QGR12. Arizona Game and Fish Department.

Chow, V. T. 1959. Open-channel hydraulics. McGraw-Hill Book, New York.

Clary, W. P. 1999. Stream channel and vegetation responses to late spring cattle grazing. Journal of Range Management 52:218-227.

Constantz, J., C. L. Thomas, and G. Zellweger. 1994. Influence of diurnal variations in stream temperature on streamflow loss and groundwater recharge. Water Resources Research 30:3253-3264.

Constantz, J. 1998. Interaction between stream temperature, streamflow, and groundwater exchanges in alpine streams. Water Resources Research 34:1609-1615.

Daufresne, M., and P. Boet. 2007. Climate change impacts on structure and diversity of fish communities in rivers. Global Change Biology 13:2467–2478.

Davies, P. E., and M. Nelson. 1994. Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Australian Journal of Marine and Freshwater Research 45:1289-1305.

Dickerson, B. R. and G. L. Vinyard. 1999. Effects of high chronic temperatures and diel temperature cycles on the survival and growth of Lahontan cutthroat trout. Transactions of the American Fisheries 128:516-521.

Dowling, T. E. and M. R. Childs. 1992. Impact of hybridization on a threatened trout of the southwestern United States. Conservation Biology 6:355-364.

Dubayah, R., and P. M. Rich. 1995. Topographic solar radiation models for GIS. International Journal of Geographic Information Science 9:405-419.

Eaglin, G. S. and W. A. Hubert. 1993. Effects of logging and roads on substrate and trout in streams of the Medicine Bow National Forest, Wyoming. North American Journal of Fisheries Management 13:844-846.

Elliott J. M. 2000. Pools as refugia for brown trout during two summer droughts: trout responses to thermal and oxygen stress. Journal of Fish Biology 56:938–948.

Ellison, D., M. N. Futter and K. Bishop. 2012. On the forest cover-water yield debate: from demand- to supply-side thinking. Global Change Biology 18:806-820.

Evans, E. C., and G. E. Petts. 1997. Hyporheic temperature patterns within riffles. Hydrological Sciences Journal 42:199-213.

Featherstone, A. W. 2012. Common or black alder. Trees for life, Forrest, Scotland. Available: www.treesforlife.org.uk/forest/species/alder.html.

Fernandez, P. J., and P. C. Rosen. 1996. A microcosm study of effects of an introduced crayfish (*Orconectes virilis*) on native Arizona aquatic plants, invertebrates, and herpetofauna from Arizona. Final Report to Arizona Game and Fish Department. Heritage IIAPM Project No. I94054.

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

Fritz, K. M., B. R. Johnson, and D. M. Walters. 2006. Field Operations Manual for Assessing the Hydrologic Permanence and Ecological Condition of Headwater Streams. EPA/600/ R-06/126. U.S. Environmental Protection Agency, Office of Research and Development, Washington DC.

Gaffield, S. J., K. W. Potter, and L. Wang. 2005. Predicting the summer temperature of small streams in southwestern Wisconsin. Journal of the American Water Resources Association 41:25-36.

Gothreaux, C. T. and C. C. Green. 2012. Effects of shading on the reproductive output and embryo viability of gulf killifish. North American Journal of Aquaculture 74:266-272.

Guoyuan Li, C. R. Jackson, and K. A. Kraseski. 2012. Modeled riparian stream shading: Agreement with field measurements and sensitivity to riparian conditions. Journal of Hydrology 428:142-151.

Hannah, D. M., I. A. Malcolm, C. Soulsby, and A. F. Youngson. 2004. Heat exchanges and temperatures within a salmon spawning stream in the Cairngorms, Scotland: Seasonal and sub-seasonal dynamics. River Research and Applications 20:635-652.

Harper, K. C. 1976. On the biology of Salmo apache and its management implications. Master's Thesis, University of Arizona, Tucson.

Harper-Smith, S. A., and Hooper, D. U. 2008. Modeling relative effects of riparian cover and groundwater inflow on stream temperature in lowland Whatcom County, Washington. 93rd Annual Ecological Society of America Meeting, Milwaukee, Wisconsin.

Hetrick, N. J., M. A. Brusven, T. C. Bjornn, R. M. Keith, and W. R. Meehan. 1998. Effects of canopy removal on invertebrates and diet of juvenile Coho Salmon in a small stream in southeast Alaska. Transaction of the American Fisheries Society 127:876-888.

Hetrick, N. J., M. A. Brusuen, W. R. Meehan, and T. C. Bjornn. 1998. Changes in solar input, water temperature, periphyton accumulation, and allochthonous input and storage after canopy removal along two small salmon streams in southeast Alaska. Transactions of the American Fisheries Society 127:859-875.

Hewlett, J. D. and J. C. Fortson. 1982. Stream temperature under an inadequate buffer strip in the southeast piedmont. Water Resources Bulletin 18:983-988.

Hibbert, A. R. 1965. Forest treatment effects on water yield. Coweeta Hydrologic Laboratory, Southeastern Forest Experiment Station, Asheville, North Carolina.

Hitt, N. P. 2003. Immediate effects of wildfire on stream temperature. Journal of Freshwater Ecology 18:171-173.

Hokanson, K. E. F., J. H. McCormick, B. R. Jones, and J. H. Tucker. Thermal requirements for maturation, spawning, and embryo survival of the brook trout, Salvelinus fontinalis. Journal of the Fisheries Research Board of Canada 3:975-984.

Inman, T. C., P. C. Marsh, B. E. Bagley, and C. A. Pacey. 1998. Survey of crayfishes of the Gila River Basin, Arizona and New Mexico, with notes on occurrences in other Arizona drainages and adjoining states. Report to U.S. Bureau of Reclamation, Phoenix, Arizona.

Isaak, D. J., C. H. Luce, B. E. Rieman, D. E. Nagal, E. E. Peterson, D. L. Horan, S. Parkes, and G. L. Chandler. 2010. Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. Ecological Applications 20:1350-1371.

Isaak, D. J, S. Wollrab, D. Horan, and G. Chandler. 2011. Climate change effects on stream and river temperatures across the northwest U.S. from 1980-2009 and implications for salmonid fishes. Climatic Change 113:499-524.

Johnson, S. L. 2004. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. Canadian Journal of Fisheries and Aquatic Sciences 61:913-923.

Karl, T. R., J. M. Melillo, and T. C. Peterson. 2009. Global climate change impacts in the United States. United States Global Climate Change Research Program. Cambridge University Press: New York, NY.

Kauffman, J. B. and W. C. Kreuger. 1984. Livestock impacts on riparian ecosystems and streamside management implication…a review. Journal of Range Management 37:430-438.

Kauffman, J. B., R. L. Beschta, N. Otting, and D. Lytjen. 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22:12-24.

Kiffney, P. M., J. S. Richardson, and J. P. Bull. 2004. Establishing light as a casual mechanism structuring stream communities in response to experimental manipulation of riparian buffer width. Journal of the North American Benthological Society 23:542-555.

Koetsier, P., Q. Tuckett, and J. White. 2007. Present effects of past wildfires on the diets of stream fish. Western North American Naturalist 67:429-438.

Kraft, M. E. 1972. Effects of controlled flow reduction on a trout stream. Journal of the Fisheries Research Board of Canada 29:1405-1411.

Larson, L. L. and S. L. Larson. 1996. Riparian shade and stream temperature: a perspective. Rangelands 18:149-152.

LeBlanc, R. T., R. D. Brown, and J. E. Fitz Gibbon. 1997. Modeling the effects of land use change on the water temperature in unregulated urban streams. Journal of Environmental Management 49:445-469.

Lee, C. G., A. P. Farrell, A. Lotto, M. J. MacNutt, S. G. Hinch, and M. C. Healey. 2003. The effect of temperature on swimming performance and oxygen consumption in adult sockeye (*Oncorhynchus nerka*) and coho (*O. kisutch*) salmon stocks. Journal of Experimental Biology 206:3239-3251.

Lee, R. M., and J. N. Rinne. 1980. Critical thermal maxima of five trout species in the southwestern United States. Transactions of the American Fisheries Society 109:632-635.

Lenat, D. R. 1984. Agriculture and stream water quality: A biological evaluation of erosion control practices. Environmental Management 8:333-343.

Lewis, S. L. 1969. Physical factors influencing fish populations in pools of a trout stream. Transactions of the American Fisheries Society 98:14-19.

Lile, D., D. Lancaster, M. Porath, J. Morrison, and Y. Sado. 2005. Statistical anaylsis of monitoring data aids in prediction of stream temperature. California Agriculture 59:161-167.

Loheide, S. No Date. Innovative methods for characterizing natural riparian discharge. Available: https://pangea.stanford.edu/research/groups/hydrogeology/research.php?rg_id=15&rgpr_id=25.

Marine, K. R., and J. J. Cech Jr. 2004. Effects of high water temperature on growth, smoltification, and predator avoidance in juvenile Sacramento River chinook salmon. North American Journal of Fisheries Management 24:198-210.

Matney, C. A. 2004. Placement of felled western juniper trees over a stream channel: effects on water temperature, streamside willow shrubs, and redband trout. Master’s Thesis. Oregon State University, Corvallis, Oregon.

Matthews, K. R. and N. H. Berg. 1997. Rainbow trout responses to water temperature and dissolved oxygen stress in two southern California stream pools. Journal of Fish Biology 50:50-67.

Mellina, E., S. G. Hinch. 2009. Influences of riparian logging and in-stream large wood removal on pool habitat and salmonid density and biomass: a meta-analysis. Canadian Journal of Forest Research 39:1280-1301.

Mesick, C. F. 1988. Effects of food and cover on numbers of Apache and brown trout establishing residency in artificial stream channels. Transactions of the American Fisheries Society 117:421-431.

Moore, R. D., D. L. Spittlehouse, and A. Story. 2005. Riparian microclimate and stream temperature response to forest harvesting: a review. Journal of the American Water Resources Association 41:813-834.

Morrill, J. C., R. C. Bales, and M. H. Conklin. 2005. Estimating stream temperature from air temperature: Implications for future water quality. Journal of Environmental Engineering 131:139-146.

Murphy, M. L., J. Heifetz, S. W. Johnson, K. V. Koski, and J. F. Thedinga. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Canadian Journal of Fisheries and Aquatic Sciences 43:1521-1533.

Nellessen, J. E. No date. *Salix exigua* Nutt. Taschek Environmental Consulting, Albuquerque, New Mexico. Available: http://www.fs.fed.us/global/iitf/pdf/shrubs/Salix%20exigua.pdf

Nielsen, J. L., T. E. Lisle, and V. Ozaki. 1994. Thermally stratified pools and their use by steelhead in Northern California streams. Transactions of the American Fisheries Society 123:613-626.

North Carolina State University. No Date. Riparian Buffers. North Carolina State University, Raleigh, North Carolina. Available: http://www.bae.ncsu.edu/programs/extension/wqg/sri/riparian5.pdf.

Oakley, A. L., J. A. Collins, L. B. Everson, D. A. Heller, J. C. Howerton, and R. E. Vincent. 1985. Riparian zones and freshwater wetlands. Management of wildlife and fish habitats in forests of western Oregon and Washington. USDA Forest Service, Portland, Oregon, Publication No. R6-F&WL-192-1985.

Opperman, J. J., K. A. Lohse, C. Brooks, N. M. Kelly, and A. M. Merenlender. 2005. Influence of land use on fine sediment in salmonid spawning gravels within the Russian River Basin, California. Canadian Journal of Aquatic Sciences 62:2740-2751.

Opti-Logic Corporation. 2004. Operating Instructions for the Laser Rangefinder/Hypsometer, Tullahoma, Tennessee.

Oregon Plan for Salmon and Watersheds. 1999. Stream Temperature Protocol, Version 2.0. Water Quality Monitoring Guidebook.

Osborne L. L., and D. A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243–58.

Packman, A. I., and M. Salehin. 2003. Relative roles of stream flow and sedimentary conditions in controlling hyporheic exchange. Hydrobiologia 494:291-297.

Peck, S. A. 1985. Effects of aggressive interaction on temperature selection by the crayfish, *Orconectes virilis*. American Midland Naturalist 114:159-167.

Platts, W. S., R. J. Torquemada, M. L. McHenry, and C. K. Graham. 1989. Changes in salmon spawning and rearing habitat from increased delivery of fine sediment to the South Fork Salmon River, Idaho. Transactions of the American Fisheries Society 118:274-283.

Poole, G. C. and C. H. Berman. 2001. An ecological perspective on in-stream temperature: Natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management 27:787-802.

Prentice, C., W. Cramer, S. P. Harrison, R. Leemans, R. A. Monserud, and A. M. Solomon. 1992. A global biome model based on plant physiology and dominance, soil properties and climate. Journal of Biogeography 19:117-134.

Quigley, J. T., and S. G. Hinch. 2006. Effects of rapid experiemental increases on acute physiological stress and behavior of stream dwelling juvenile chinook salmon. Journal of Thermal Biology 31:429-441.

Quinn, J. M., and A. E. Wright-Stow. 2008. Stream size influences stream temperature impacts and recovery rates after clearfell logging. Forest Ecology and Management 256:2101-2109.

Recsetar, M. S. 2011. Thermal tolerance of Apache trout at various life stages. Master’s thesis. The University of Arizona, Tucson, Arizona.

Recsetar, M. S., M. P. Zeigler, D. L. Ward, S. A. Bonar, and C. A. Caldwell. 2012. Relationship between fish size and upper thermal tolerance. Transactions of the American Fisheries Society 141:1433-1438.

Richardson, D. M., P. M. Holmes, K. J. Esler, S. M. Galatowitsch, J. C. Stromberg, S. P. Kirkman, P. Pysek, and R. J. Hobbs. 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Diversity and Distributions 13:126-139.

Rinaldi, M., B. Wyzga, and N. Surian. 2005. Sediment mining in alluvial channels: physical effects and management perspectives. River Research and Applications 21:805-828.

Rinne, J. N. and W. L. Minckley. 1985. Patterns of variation and distribution in Apache trout (*Salmo apache*) relative to co-occurrence with introduced salmonids. Copeia 1985:285-292.

Rinne, J. N. and J. Janisch. 1995. Coldwater fish stocking and native fishes in Arizona: past, present, and future. American Fisheries Society Symposium 15: 397-406.

Robinson, A. T., L. D. Avenetti, and C. Cantrell. 2004. Evaluation of Apache trout habitat protection actions. Arizona Game and Fish Department, Research Branch, Technical Guidance Bulletin No. 7, Phoenix, Arizona.

Rosgen, D. L. 1997. A geomorphological approach to restoration of incised rivers. Proceedings of the Conference on Management of Landscapes Disturbed by Channel Incision. Center for Computational Hydroscience and Bioengineering 3-22, Oxford Campus, University of Mississippi.

Rowe, P. B. 1963. Streamflow increases after removing woodland-riparian vegetation from a southern California watershed. Journal of Forestry 61:365-370.

Roulet, N. T. 1990. Hydrology of a headwater basin wetland: Groundwater discharge and wetland maintenance. Hydrological Processes 4:387-400.

Ruochuan G., S. Montgomery, and T. A. Austin. 1998. Quantifying the effects of stream discharge on summer river temperature. Hydrological Sciences 43:885-904.

Rutherford, J. C., S. Blackett, C. Blackett, L. Saito, and R. J. Davies-Colley. 1997. Predicting the effects of shade on water temperature in small streams. New Zealand Journal of Marine and Freshwater Research 31:707-721.

Sidle, R. C., and A. Sharma. 1996. Stream channel changes associated with mining and grazing the Great Basin. Journal of Environmental Quality 25:111-1121.

Sinokrot, B. A., and H. G. Stefan. 1993. Stream temperature dynamics: measurements and modeling. Water Resources Research 29:2299-2312.

Sinokrot, B. A., and H. G. Stefan. 1994. Stream water-temperature sensitivity to weather and bed parameters. Journal of Hydraulic Engineering 120:722-736.

Sovell, L. A., B. Vondracek, J. A. Frost, and K. G. Mumford. 2000. Impacts of rotational grazing and riparian buffers on physicochemical and biological characteristics of southeastern Minnesota, USA, streams. Environmental Management 26:629-641.

Steed, J. E., and L. E. DeWald. 2003. Transplanting sedges (*Carex* spp.) in southwestern riparian meadows. Restoration Ecology 11:247-256.

Storck, P., L. Bowling, P. Wetherbee, and D. Lettenmaier. 1998. Application of a GIS-based distributed hydrology model for prediction of forest harvest effects on peak stream flow in the Pacific Northwest. Hydrological Processes 12:889-904.

Sugimoto, S., F. Nakamura, and A. Ito. 1997. Heat Budget and statistical analysis of the relationship between stream temperatures and riparian forest in the Toikanbetsu River Basin, northern Japan. Canadian Journal of Forest Research 2:103-107.

Suttle, K. B., M. E. Power, J. M. Levine, and C. McNeely. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecological Applications 14:969-974.

Swank, W. T., J. M. Vose, and K. J. Elliott. 2001. Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a southern Appalachian catchment. Forest Ecology and Management 143:163-178.

Swift Jr., L. W., and W. T. Swank. 1981. Long term responses of streamflow following clearcutting and regrowth. Hydrological Sciences Bulletin 26:245-256.

Tate, K. W., D. L. Lancaster, and D. F. Lile. 2007. Assessment of thermal stratification within stream pools as a mechanism to provide refugia for native trout in hot, arid, rangelands. Environmental Monitoring and Assessment 124:289-300.

Theurer, F. D., K. A. Voos, and W. J. Miller. 1984. Instream water temperature model. Instream Flow Information Paper 16. United States Fish and Wildlife Service. FWS/OBS-85/15. v.p.

Theurer, F. D., I. Lines, and T. Nelson. 1985. Interaction between riparian vegetation, water temperature, and salmonid habitat in the Tucannon River. Water Resources Bulletin 21: 53-64.

Trimble, S. W. 1994. Erosional effects of cattle on streambanks in Tennessee, U.S.A. Earth Surface Processes and Landforms 19:451-464.

Trimble, S. W. 1997. Stream channel erosion and change resulting from riparian forests. Geology 25:467-469.

USDA, NRCS. 2013. The PLANTS Database. National Plant Data Team, Greensboro, North Carolina. Available: http://plants.usda.gov.

USFWS (U. S. Fish and Wildlife Service). 2009. Apache Trout Recovery Plan, Second Revision. Albuquerque, New Mexico.

Walser, C. A., and H. L. Bart. 1999. Influence of agriculture on in-stream habitat and fish community structure in Piedmont watersheds of the Chattahoochee River system. Ecology of Freshwater Fish 8:237-246.

Ward, J. V. 1985. Thermal characteristics of running waters. Hydrobiologia 125:31-46.

Wenger, S. J., D. J. Isaak, C. H. Luce, H. M. Neville, K. D. Fausch, J. B. Dunham, D. C. Dauwalter, M. K. Young, M. M. Elsner, B. E. Rieman, A. F. Hamlet, and J. E. Williams. 2011. Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. Proceedings of the National Academy of Sciences 108:14175–14180.

Western Regional Climate Center. No date. Period of Record Monthly Climate Summery. Western Regional Climate Center, Greer, Arizona. Available: http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?az3683. (June 2012).

Whitledge, G. W., C. F. Rabeni, G. Annis, and S. P. Sowa. 2006. Riparian shading and groundwater enhance growth potential for smallmouth bass in Ozark streams. Ecological Applications 16:1461-1473.

Williams, J. E., and J. Meka Carter. 2009. Managing native trout past peak water. Southwest Hydrology 8:26-27.

Wilzbach, M. A. 1985. Relative roles of food abundance and cover in determining the habitat distribution of stream-dwelling cutthroat trout (*Salmo clarki*). Canadian Journal of Fisheries and Aquatic Sciences 42:1668-1672.

Winter, T. C. 1999. Relation of streams, lakes, and wetlands to groundwater flow systems. Hydrology Journal 7:28-45.

Wu, F. 2000. Modeling embryo survival affected by sediment deposition into salmonid spawning gravels: application to flushing flow prescriptions. Water Resources Research 36:1595-1603.

Xu, C. L., B. H. Letcher, and K. H. Nislow. 2010. Site-dependent survival of brook trout *Salvelinus fontinalis* in summer: effects of water temperature and stream flow. Journal of Fish Biology 76:2342-2369.

Yoshida, K., H. Tantji, H. Somura, O. Toda, and T. Masumoto. 2004. Evaluation of stream temperature change with reservoir management in the Nam Ngum basin. Proceedings of ICID 2nd Asian Regional Conference in Australia 3:1-8.

Table 1 - Sampled average upstream water temperature, estimated average upstream water temperature under average maximum air temperature conditions over a 12-yr period, and average upstream water temperature when a 6 °C increase in maximum annual air temperature due to climate change is added.

stream	sampled upstream average temperature (°C)	upstream water temperature under maximum annual air temperature conditions (°C)	upstream water temperature under a global climate change scenario (°C)
Hayground Creek	8.637	16.035	23.433
Conklin Creek	11.202	14.770	18.338
West Fork Black River	9.883	13.601	17.319
West Fork Little Colorado River	6.575	10.233	13.891

Table 2 - Segment Inflow and outflow for each stream modeled.

Stream	Segment Inflow (cms)	Segment Outflow (cms)
Hayground Creek	0.005	0.002
Conklin Creek	0.013	0.007
West Fork Black River	0.677	0.468
West Fork Little Colorado River	0.385	0.478

Table 3 - A and b terms of width calculated from the natural log relationship between the channel wetted width and discharge, for each stream modeled.

Stream	Width's a term	Width's b term
Hayground Creek	1.071	.015
Conklin Creek	2.493	.052
West Fork Black River	3.894	.561
West Fork Little Colorado River	3.781	.098

Table 4 - Meteorological data collected on-site at modeled White Mountain stream segments using portable weather station, May 16-20. This information was used to calibrate the model to each site.

Stream Segment	Mean Daily Air Temperature (°C)	Maximum Air Temperature (°C)	Mean Daily Relative Humidity (%)	Mean Daily Wind Speed (mps)	Mean Daily Solar Radiation (langleys/day)
Hayground Creek	7.41	17.7	37.79	3.78	367.31
Conklin Creek	14.16	25.84	33.58	0.36	285.2
West Fork Black River	7.55	19.29	42.08	1.19	326.62
West Fork Little Colorado River	8.6	19.3	34.3	0.44	321.5

Table 5 - Vegetation characteristics for each stream modeled.

Stream	Segment Azimuth (radians)	Mean Topographic Altitude (radians)		Mean Vegetation Height (m)		Mean Vegetation Crown (m)		Mean Vegetation Offset (m)		Total Vegetation Density (%)	
		West	East	West	East	West	East	West	East	West	East
Hayground Creek	1.480	0.314	0.381	16.072	17.879	5.374	6.207	12.325	19.492	25.923	39.355
SE		0.056	0.051	0.831	2.032	0.497	0.673	4.107	3.513		
Conklin Creek	-1.190	0.649	0.602	22.053	20.565	7.212	7.418	7.727	8.032	74.079	64.710
SE		0.029	0.015	2.093	2.018	0.523	0.502	1.279	1.084		
West Fork Black River	-0.785	0.120	0.134	16.028	20.589	6.313	7.171	3.200	24.714	7.998	7.617
SE		0.019	0.022	2.950	0.850	2.200	0.400	1.600	5.400		
West Fork Little Colorado River	0.870	0.280	0.200	14.135	13.284	6.160	5.723	5.428	6.400	64.618	74.401
SE		0.022	0.033	0.958	0.850	0.365	0.286	1.123	0.901		

Table 6 - Vegetation density, by species, in percent, for each stream modeled.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	coyote Willow	Other	Unvegetated
Hayground Creek	0	30	10	0	0	0	0	60
Conklin Creek	47	3	32	5	0	1.5	10	1.5
West Fork Black River	0	9	0	0	0	0	0	91
West Fork Little Colorado River	0	88	0	0	2	8	0	2

Table 7 - Mean Vegetation characteristics for the six most common riparian vegetation species along the streams sampled.

Stream	Mean Height (m)	Mean Crown (m)	Mean Density, quality (%)
Douglas fir	24.362 (SE 1.446)	7.818 (SE 0.371)	82.40 (SE 0.034)
Engelmann spruce	17.572 (SE 0.594)	7.179 (SE 0.233)	78.788 (SE 0.032)
Ponderosa pine	24.450 (SE 1.103)	8.833 (SE 0.293)	76.973 (SE 0.033)
Arizona alder	7.673 (SE 0.570)	5.583 (SE 0.332)	77.640 (SE 0.034)
Bebb willow	6.353 (SE 0.765)	5.808 (SE 0.473)	78.131 (SE 0.039)
coyote willow	3.027 (SE 0.317)	4.106 (SE 0.559)	67.615 (SE 0.089)

Table 8 - Predicted maximum downstream temperature, sampled maximum downstream temperature, estimate of maximum downstream temperature when maximum annual air temperature is modeled, and estimated maximum downstream temperature when a 6 ^{0}C increase in maximum annual air temperature due to climate change is modeled.

Stream	Predicted Maximum Downstream Temp (°C)	Sampled Maximum Downstream Temp (°C)	Estimated downstream Maximum Temperature with Maximum Annual Air Temperature (°C)	Estimated Downstream Maximum Temperature Under a Climate Change Scenario (°C)
Hayground Creek	25.39	25.32	28.52	31.12
Conklin Creek	24.95	25.61	30.36	33.14
West Fork Black River	19.25	19.19	22.42	25.39
West Fork Little Colorado River	16.23	16.51	20.95	24.23

Table 9 - Segment length, measured rate of change in water temperature from upstream point to downstream point of segment, mean width, mean discharge, and total shade, for each stream modeled.

	Segment Length (m)	Rate of change in water temperature (°C/m)	Mean Width (m)	Mean Discharge (cms)	Total Shade (%)	Groundwater Input (cms)
Hayground Creek	2000	0.0016	0.982 (SE 0.147)	0.003 (SE 0.001)	1.873	-0.003
Conklin Creek	3245	0.0014	2.1 (SE 0.192)	0.038 (SE 0.006)	48.954	-0.006
West Fork Black River	2215	0.00045	2.73 (SE 0.195)	0.524 (SE 0.029)	5.483	0.161
West Fork Little Colorado River	5056	0.00039	3.566 (SE 0.137)	0.55 (SE 0.026)	41.376	0.093

Table 10 - Increase in stream discharge, in cms, required to lower annual maximum downstream temperature 1 °C and below the LT_{50} for Apache trout, before and after a 6 °C increase in mean air temperature. "-" indicates that temperature goals were unobtainable by increasing discharge. "NA" indicates that maximum downstream temperature of the stream segment did not exceed the Apache trout LT_{50} under current climate conditions.

Stream	1°C	Below LT_{50} for Apache trout	Below LT_{50} after 6°C increase in air temperature
Hayground Creek	0.042	0.472	-
Conklin Creek	0.038	0.626	5.071
West Fork Black River	0.303	NA	-
West Fork Little Colorado River	0.201	NA	0.362

Table 11 - Increase in groundwater input, in cms and (percent increase), within segment, required to lower annual maximum downstream water temperature 1 °C and below the LT_{50} for Apache trout both before and after a 6 °C increase in mean air temperature. “NA” indicates that maximum downstream temperature of the stream segment did not exceed the Apache trout LT_{50} under current climate conditions.

Stream	1 C	LT_{50}	Climate Change
Hayground Creek	0.019 (950%)	0.088 (4400%)	0.134 (6700%)
Conklin Creek	0.02 (286%)	0.143 (2043%)	0.213 (3043%)
West Fork Black River	0.138 (16%)	NA	0.313 (37%)
West Fork Little Colorado River	0.056 (12%)	NA	0.068 (14%)

Table 12 - Decrease in stream width, in m, required to lower annual maximum downstream temperature 1 °C and below the LT_{50} for Apache trout, before and after a 6 °C increase in mean air temperature. “-“ indicates that temperature goals were unobtainable by increasing stream width. “NA” indicates that maximum downstream temperature of the stream segment did not exceed the Apache trout LT_{50} under current climate conditions.

Stream	1°C	Below LT_{50} for Apache trout	Below LT_{50} after a 6° C increase in air temperature
Hayground Creek	-	-	-
Conklin Creek	1.395	-	-
West Fork Black River	0.602	NA	2.303
West Fork Little Colorado River	0.559	NA	0.769

Table 13 - Percent increase in total shade required to lower annual maximum downstream temperature 1 ^{0}C and below the LT_{50} for Apache trout, before and after a 6 ^{0}C increase in mean air temperature due to climate change. "NA" indicates that maximum downstream temperature of the stream segment did not exceed the Apache trout LT_{50} under current climate conditions.

Stream	1 °C	Below LT_{50} for Apache trout	Below LT_{50} after a 6 °C increase in air temperature
Hayground Creek	9.703	50.345	76.607
Conklin Creek	5.404	36.325	51.871
West Fork Black River	10.230	NA	27.658
West Fork Little Colorado River	6.181	NA	9.080

Table 14 - Additional number of trees, per 100 meters of stream, needed to lower downstream temperature to 1 ^{o}C, for each stream modeled.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	7.9	8.9	7	11.9	11.7	16.6
Conklin Creek	0.6	0.7	0.6	1.0	1.0	1.9
West Fork Black River	1.2	1.4	1.1	2.1	2.0	3.7
West Fork Little Colorado River	0.6	0.7	0.5	1.0	1.0	1.6

Table1 5: Additional number of trees, by species, needed to lower maximum downstream water temperature estimate 1 ^{o}C, for each stream modeled. Within rows, different letters denote significant differences.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	156 z	174 z y	138 z	234 x	229 y x	327 w
Conklin Creek	39 z	48 z y	40 z	71 x	68 y x	127 w
West Fork Black River	74 z	86 z y	68 z	126 x	122 y x	226 w
West Fork Little Colorado River	103 z	127 z y	93 z	180 x	175 y x	291 w

Table 16 - Additional number of trees, per 100 meters of stream, needed to lower downstream temperature below the LT_{50} for Apache trout, for each stream modeled. "-" indicates that temperature goals were unobtainable by adding specific species of tree to model.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	8.9	9.8	7.9	11.1	12.4	18.6
Conklin Creek	5.5	6.8	5.2	9.9	9.4	-
West Fork Black River	NA	NA	NA	NA	NA	NA
West Fork Little Colorado River	NA	NA	NA	NA	NA	NA

Table 17 - Additional number of trees, per 100 meters of stream, needed to lower downstream temperature below the LT_{50} for Apache trout, after a 6 °C increase in average air temperature due to climate change. "-" indicates that temperature goals were unobtainable by adding specific species of tree to model.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	21.6	27.0	22.3	-	-	-
Conklin Creek	-	-	-	-	-	-
West Fork Black River	2.8	3.2	2.6	4.5	4.4	8.0
West Fork Little Colorado River	0.8	0.9	0.7	1.2	1.2	2.2

Table 18 - Decrease in stream depth, in mm per day, due to the addition of trees, by species, needed to lower annual maximum downstream temperature 1 °C.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	1.671	1.572	1.887	0.799	0.846	0.604
Conklin Creek	0.059	0.061	0.077	0.034	0.035	0.033
West Fork Black River	0.067	0.065	0.078	0.036	0.038	0.035
West Fork Little Colorado River	0.086	0.090	0.099	0.048	0.051	0.042

Table 19 - Decrease in stream depth, in mm per day, due to the addition of trees, by species, needed to lower current annual maximum downstream temperature below the Apache trout LT_{50}. "-" indicates that temperature goals were unobtainable by adding specific species of tree to model. "NA" indicates that maximum downstream temperature of the stream segment did not exceed the Apache trout LT_{50} under current climate conditions.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	1.875	1.743	2.133	0.744	0.898	0.676
Conklin Creek	0.569	0.590	0.678	0.323	0.334	-
West Fork Black River	NA	NA	NA	NA	NA	NA
West Fork Little Colorado River	NA	NA	NA	NA	NA	NA

Table 20 - Decrease in stream depth, in mm per day, due to the addition of trees, by species, needed to lower annual maximum downstream temperature below the Apache trout LT_{50}, after a 6 ^{0}C increase in mean air temperature due to climate change. "-" indicates that temperature goals were unobtainable by adding specific species of tree to model.

Stream	Douglas Fir	Engelmann Spruce	Ponderosa Pine	Arizona Alder	Bebb Willow	Coyote Willow
Hayground Creek	4.542	4.787	5.976	-	-	-
Conklin Creek	-	-	-	-	-	-
West Fork Black River	0.346	0.334	0.415	0.170	0.177	0.154
West Fork Little Colorado River	0.118	0.112	0.136	0.059	0.064	0.056

Table 21 - Additional amount of stream, in m², that needs to be totally shaded to lower annual maximum downstream temperature 1 ^{0}C and below the LT_{50} for Apache trout, before and after a 6 ^{0}C increase in mean air temperature due to climate change. "NA" indicates that maximum downstream temperature of the stream segment did not exceed the Apache trout LT_{50} under current climate conditions.

Stream	1° C	Below LT_{50} for Apache trout	Below LT_{50} after a 6° C increase in air temperature
Hayground Creek	191	989	1505
Conklin Creek	368	2475	3535
West Fork Black River	619	NA	1672
West Fork Little Colorado River	1114	NA	1637

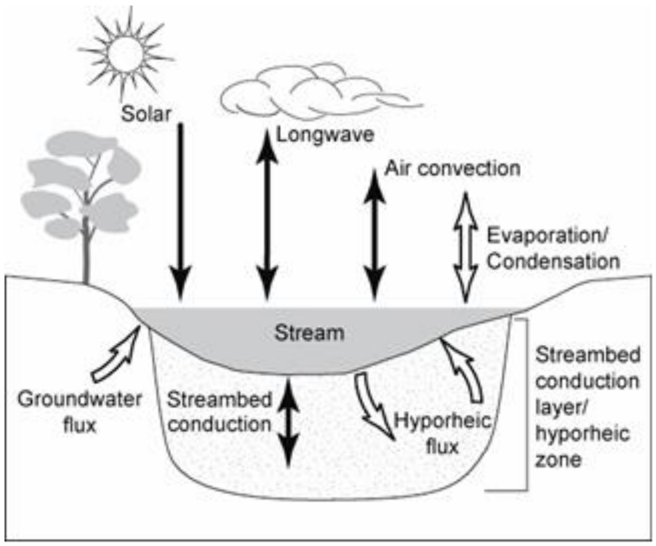
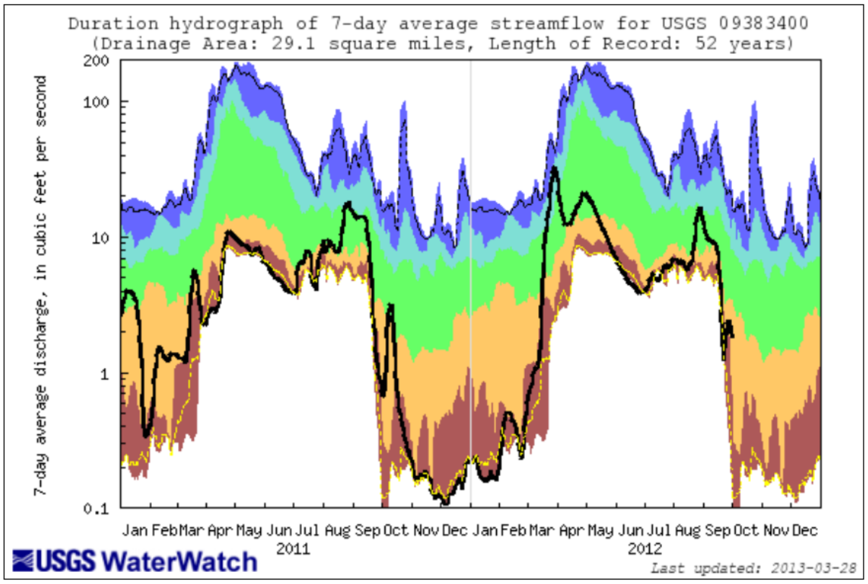
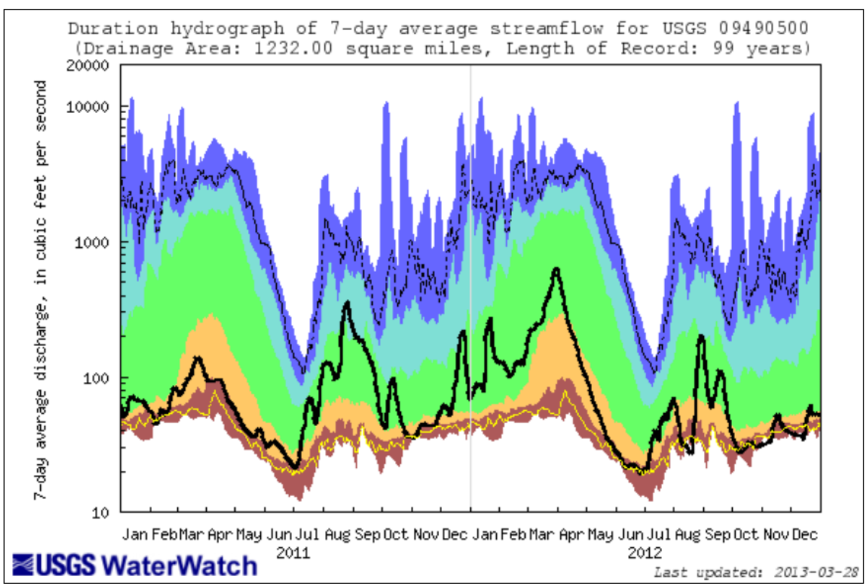


Figure 1, Factors controlling stream temperature, arrows depict energy moving into and out of the stream (Steven Loheide n.d.).





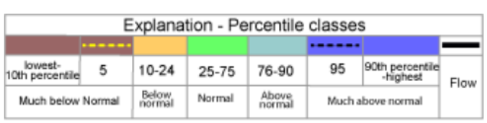


Figure 2, Hydrograph of Little Colorado River near Greer, Arizona (top) and Black River near Whiteriver, Arizona (bottom) during study period. Flow during May 2011 and 2012 was in the lower 10% ever recorded for the area. Data from USGS stream gauges 09383400 and 09490500 (USGS unpublished data).

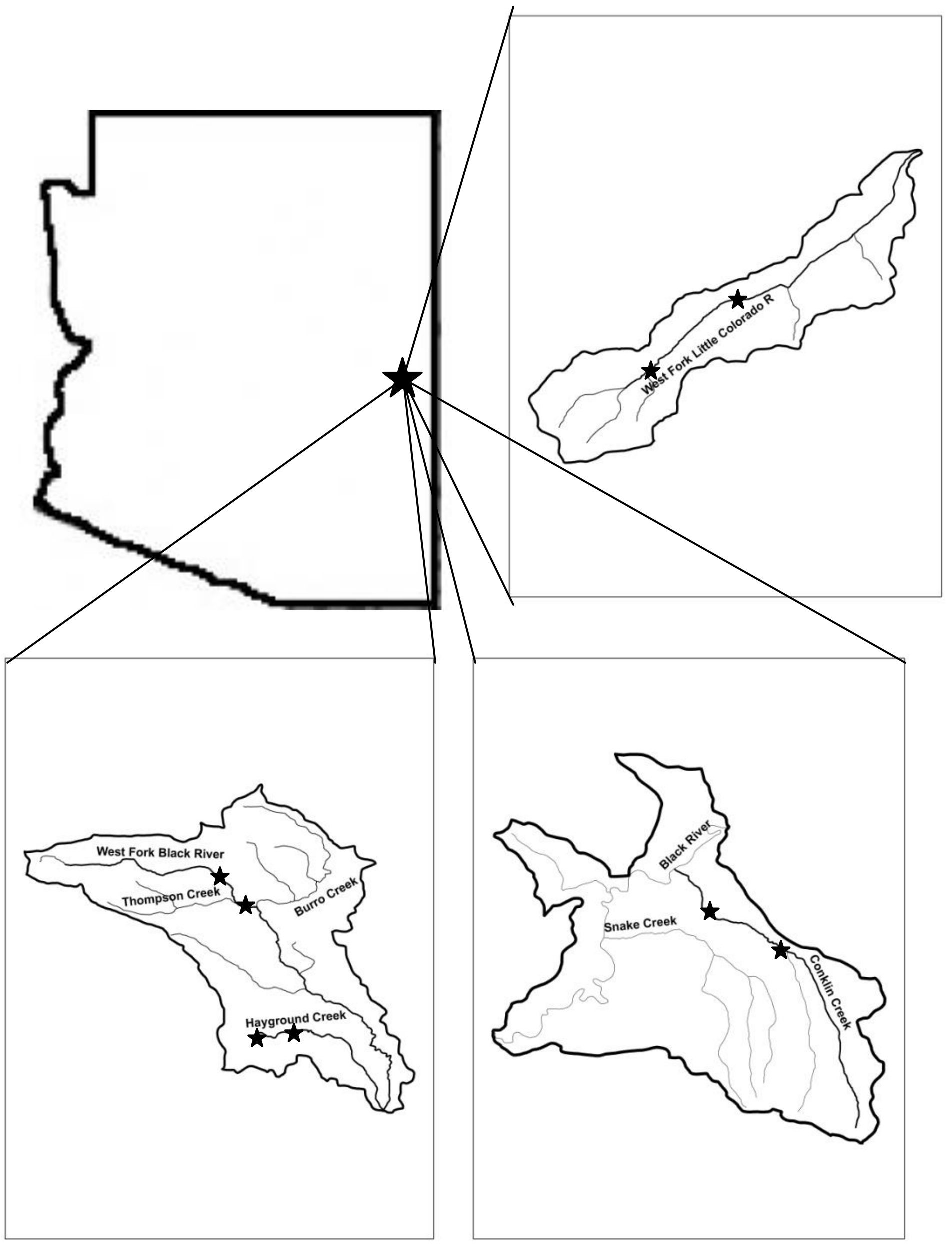
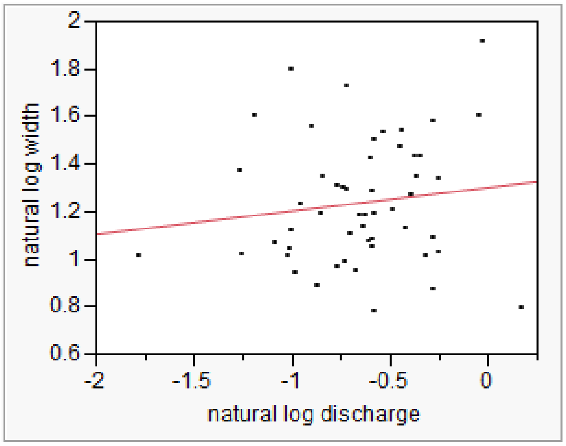


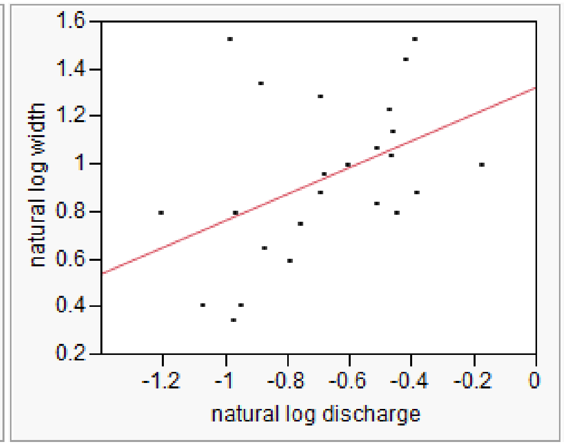
Figure 3, Map of Arizona with Apache-Sitgreaves National Forest marked with a star. Maps of the three watersheds, within the Apache-Sitgreaves National Forest, where West Fork Little Colorado River, West Fork Black River, Hayground Creek, and Conklin Creek are located; upstream and downstream points are marked with stars.

West Fork Little Colorado River



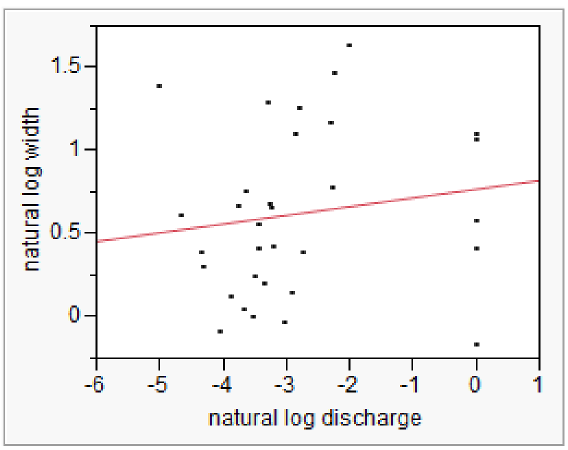
y = 1.3013696 + 0.0979697(x); r^2 = 0.017

West Fork Black River



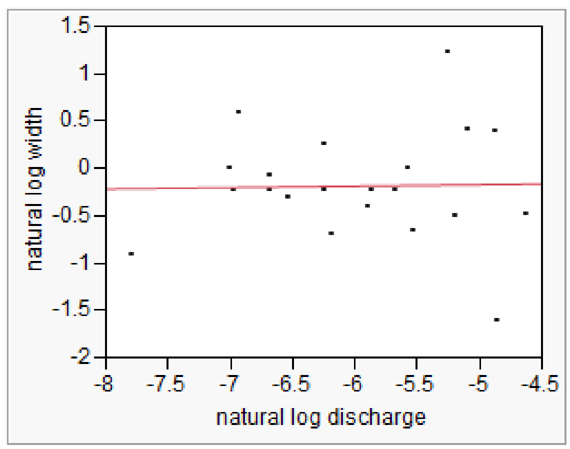
y = 1.3233083 + 0.5605599(x); r^2 = 0.192

Conklin Creek



y = .7683292 + 0.052403(x); r^2 = 0.022

Hayground Creek



y = -.096844 + 0.0149134(x); r^2 = 0.0005

Figure 4, Natural log relationship between channel wetted width and discharge for each stream sampled.

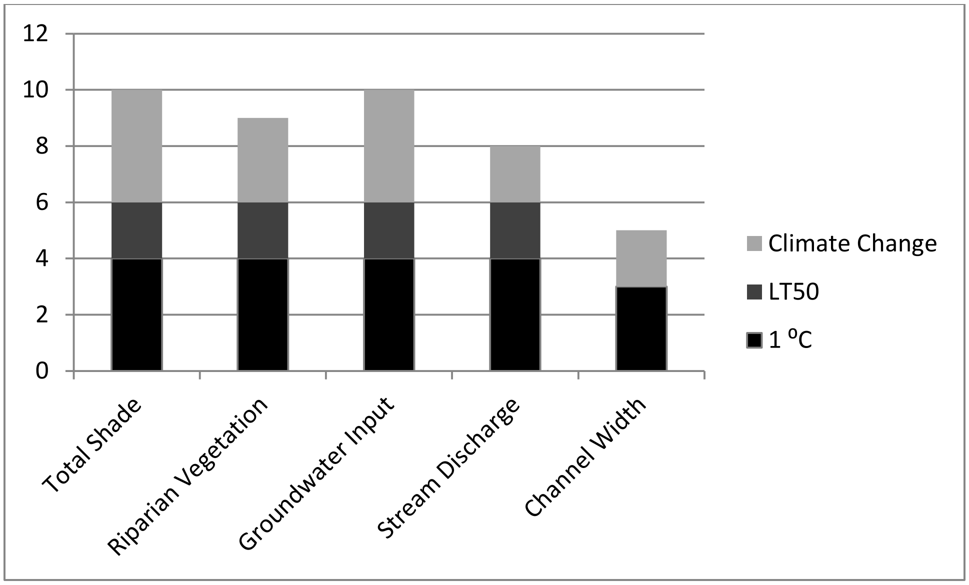


Figure 5, Frequency of successful model simulations for each cooling method.

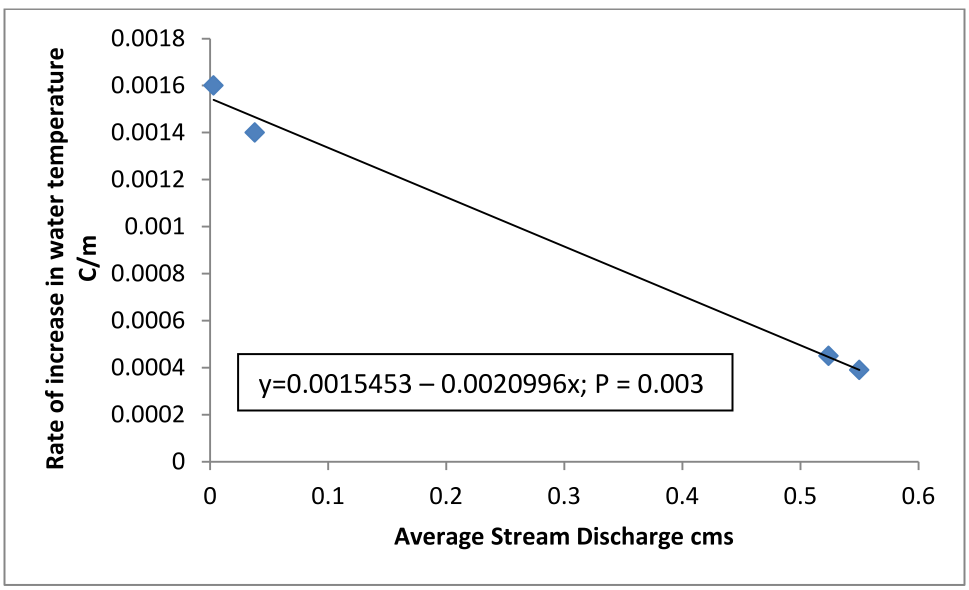


Figure 6, Relationship between average stream discharge, in cms, and the rate of increase in water temperature over distance, in ^{o}C/m.

Appendix 1. Aerial images of streams used in Apache trout temperature tolerance study.

Figure A-1. Aerial view of stream segment on West Fork Little Colorado River. Upstream point of segment is marked with a red star and downstream point is marked with a red bar.

Figure A-2, Aerial view of stream segment on West Fork Black River. Upstream point of segment is marked with a red star and downstream point is marked with a red bar.

Figure A-3, Aerial view of stream segment on Conklin Creek. Upstream point of segment is marked with a red star and downstream point is marked with a red bar.

Figure A-4, Aerial view of stream segment on Hayground Creek. Upstream point of segment is marked with a red star and downstream point is marked with a red bar.

Appendix 2. Detailed description of SSTEMP model inputs used for Apache trout research.

Model Inputs

Each stream segment begins with an upstream point, chosen based on limitations of the model or the reservation boundary and ends with a downstream point, also chosen based on limitation of the model. Starting at the upstream point, transects were placed every 100 meters. The following are detailed explanations of each model input.

Time of Year

Month/day (mm/dd) – This is the month and day I sampled the stream segment. On streams where I started on one day and ended on another, I used the day that the stream temperature loggers and weather station were out longest. For example, if I set up the temperature loggers and weather station on 5/20 at 10:00 and removed them on 5/21 at 10:00 than I used 5/20 as the date.

Hydrology

Segment Inflow (cms) – At the upstream point of the segment I measured the wetted width of the stream in m. I measured the depth of the stream ¼ distance from the bank, ½ distance from the bank, and ¾ distance from the bank and averaged the values to get one depth. I multiplied stream width by average stream depth to get stream area in m^2. I used a USGS pygmy current meter to measure stream velocity, in m/s, ¼ distance from the bank, ½ distance from the bank, and ¾ distance from the bank and averaged the values to get one measure of stream velocity. I multiplied stream area by average stream velocity to get the model input for segment inflow, in cms.

Inflow Temperature (°C) – I attached two HOBO pendant temperature/light loggers to a rebar with zip-ties and hammered the rebar into the stream at the upstream point of the segment, in the middle of the stream, in the middle of the water column, not in contact with any heat sinks or sources. The loggers were set to record temperature, in °C, every 15 minutes for 24 hours. I calculated the daily average from each logger and then averaged the two measurements to get the model input for inflow temperature, in °C.

Segment Outflow (cms) – At the downstream point of the segment I measured the wetted width of the stream in m. I measured the depth of the stream ¼ distance from the bank, ½ distance from the bank, and ¾ distance from the bank and averaged the values to get one depth. I multiplied stream width by average stream depth to get stream area in m^2. I used a USGS pygmy current meter to measure stream velocity, in m/s, ¼ distance from the bank, ½ distance from the bank, and ¾ distance from the bank and averaged the values to get one measure of stream velocity. I multiplied stream area by average stream velocity to get the model input for segment outflow, in cms.

Accretion Temperature (°C) – This is a measure of groundwater temperature. Because I had no way of directly measuring groundwater temperature, I used the mean annual air temperature, in °C. I obtained this information from the Western Regional Climate center.

Geometry

Latitude (radians) – I used a Garmin GPSMAP 60CSx to obtain latitude, in radians, of the stream segment.

Segment Length (km) – This is the distance, in km, from the upstream point to the downstream point. I used the Garmin GPSMAP 60CSx to measure the distance as I walked along the stream, taking into account stream sinuosity.

Upstream Elevation (m) – I used the Garmin GPSMAP 60CSx to calculate the elevation at the upstream point of the segment, in m.

Downstream Elevation (m) – I used the Garmin GPSMAP 60CSx to calculate the elevation at the downstream point of the segment, in m.

Width's A Term (s/m^2) and B Term where W = A*Q**B – At the upstream and downstream ends of the segment and at each transect in between, I measured the following:

Stream wetted width – I used a meter stick or the Opti-Logic 1000LH Laser Range Finder Hypsometer to measure the wetted width of the stream, in m.

Depth of the stream – I used a meter stick to measure the stream depth, in m, ¼ distance from the bank, ½ distance from the bank, and ¾ distance from the bank and averaged the three values to get one depth.

Area of channel – I multiplied stream width by average stream depth to get stream area in m^2.

Stream velocity - I used a USGS pygmy current meter to measure stream velocity, in m/s, ¼ distance from the bank, ½ distance from the bank, and ¾ distance from the bank and averaged the values to get one measure of stream velocity.

Stream discharge – I multiplied the area of the channel by the average stream velocity to get stream discharge, in m^3/s.

B Term - I plotted the width measurements I calculated at each transect on the y-axis and the discharge measurements I calculated at each transect on the x-axis. The relationship between the two should approximate a straight line and the slope of that line is the B Term.

A Term – This is calculated from the above equation, where W = the average width of the stream, Q = the average discharge of the stream segment, and B = the B Term.

Manning's n – This is a measure of the roughness of the streambed that causes water to slow due to friction. Streams of similar size and with the same bottom substrate have similar manning's n values. I used Chow (1959) to estimate manning's n for each stream segment based on size and substrate.

Meteorology

I measured most meteorological data using a HOBO Micro Station Data Logger with attached smart sensors. The weather station was set up adjacent to the stream segment I was studying, outside of the riparian area and measured air temperature, relative humidity, wind speed, and solar radiation:

Air Temperature (°C), Maximum Air Temp (°C), and Relative Humidity (%) – I attached a 12-bit temperature and relative humidity sensor to the HOBO Micro Station Data Logger and covered it with a solar radiation shield. I set the sensor to log every 15 minutes for the same 24 hour period that the water temperature sensors logged. I calculated average daily air temperature, maximum daily air temperature, and average daily relative humidity.

Wind Speed (mps) – I attached a Wind Speed Smart Sensor to the HOBO Micro Station Data Logger and set it to log every 15 minutes for the same 24 hour period that the water temperature sensors logged.

Solar Radiation ($j/m^2/s$) – I attached a solar radiation sensor (silicon pyranometer) to the HOBO Micro Station Data Logger and set it to log every 15 minutes for the same 24 hour period that the water temperature sensors logged.

Ground Temperature (°C) - Using mean annual air temperature is recommended.

Thermal Gradient ($j/m^2/s/°C$) – This parameter is often difficult to measure in the field and is not particularly sensitive within a narrow range. The author of the model recommends using the default value of 1.65 if you do not measure it in the field.

Possible Sun (percent) – This is an indirect measure of cloud cover. I obtained this value from the MSEOWEST weather station.

Dust Coefficient (dimensionless) and Ground Reflectivity (percent) – If you enter a value for solar radiation, than SSTEMP will ignore the dust coefficient and ground reflectivity.

Shade

Total Shade (%) – This is the percentage of the stream that is shaded from solar radiation. In my study, this number is based on the optional shading parameters.

Optional Shading Variables

Segment Azimuth (radians) – This is the general orientation of the stream with respect to due south. It determines which sides of the stream will be called east and west when measuring the optional shading variables. I used a TOPO map to estimate this variable and converted degrees to radians.

I measured the following shading variables at the upstream and downstream points of the segment and at each transect in between, on both the east and west sides of the stream.

Topographic Altitude (radians) – This is the average line of site angle to the horizon. I used the Opti-Logic 1000LH Laser Range Finder Hypsometer which has a vertical angle sensor that measures the angle to the horizon. I measured this from the middle of the stream.

Vegetation Height (m) – This is the average height of the shading vegetation. This is V_h in Figure 10. This includes bank height. I measured this with the Opti-Logic 1000LH Laser Range Finder Hypsometer. The vertical angle sensor allows for height measurements of objects by calculating the distance to the bottom of the object, distance to the top of the object, and the angle between the two measurements. I measured the height of the tree that would provide the stream with the most shade throughout the day.

Vegetation Crown (m) – This is the average width of the shading trees from the tip of a branch on one side to the tip of a branch on the other side. This is V_c in Figure 10. If the tree was small I used a tape measure and if the tree was large I used the Opti-Logic 1000LH Laser Range Finder Hypsometer to calculate the width of the tree.

Vegetation Offset (m) – This is the average distance to the trunks of the shading vegetation from the water's edge. This is V_o in Figure 10. If the distance was less than or equal to 3.5 meters than I used a tape measure, if the distance was greater than 3.5 meters than I used the Opti-Logic 1000LH Laser Range Finder Hypsometer.

Vegetation Density (%) – This is the average screening factor of the shade producing vegetation along the entire stream. This is V_d in Figure 10. This variable is broken up into two measurements, density quantity and density quality. Density quantity is the percentage of the stream with shading riparian vegetation; this is any vegetation closer than 35 m. I found this number by simply calculating the percentage of transects that actually had riparian vegetation closer than 35 m; this was done on the east side and west side of the stream. The second part of vegetation density is density quality and this is measure of the percentage of light filtered by the leaves and trunks of the shading vegetation. I calculated this by taking a lux reading in full sunlight using a Extech Lux Meter, model 401025 and then taking a lux reading under the tree that was shading the stream and then calculating the percentage of light filtered. I averaged the density quality measurements of each transect and multiplied it by the density quantity measurement for the corresponding side of the stream and found vegetation density for each side of the stream segment.

I also recorded the species of shade vegetation at each transect to calculate the average shading variables for each species. This information was used when I added vegetation to the model to cool the stream.

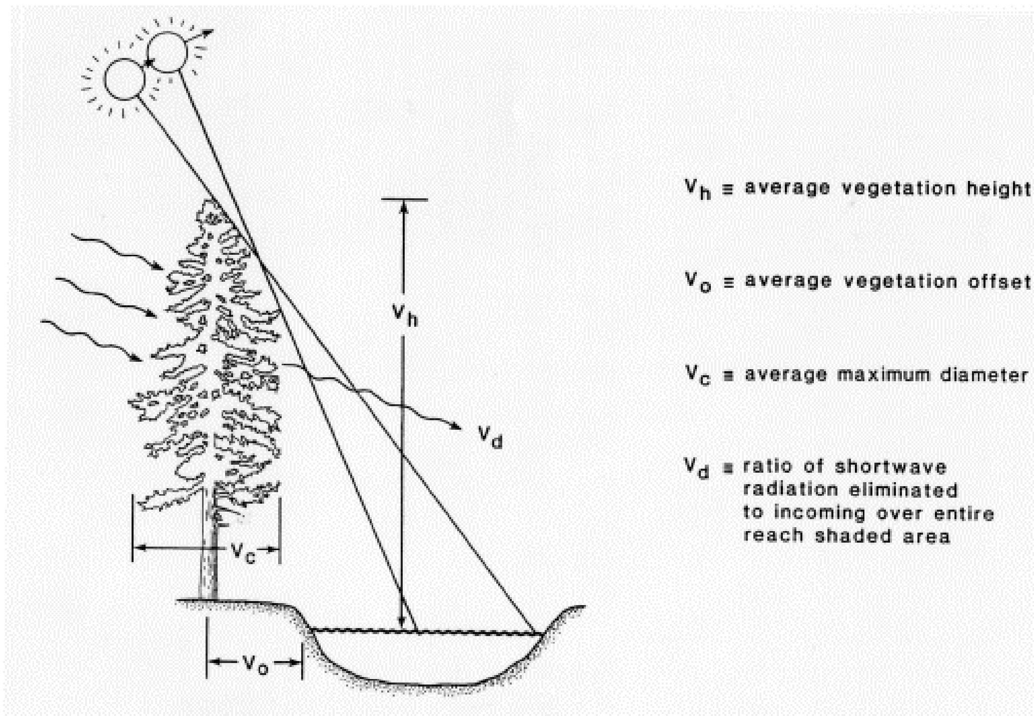


Figure A-5, Diagram of optional shading parameters for SSTEMP, from Bartholow (2002).