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Using a Mechanistic Model to Develop Management Strategies to Cool Apache Trout Streams under the Threat of Climate Change

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Abstract

User-friendly stream temperature models populated with on-site data may help in developing strategies to manage temperatures of individual stream reaches that are subject to climate change. We used the field-tested Stream Segment Temperature model (U.S. Geological Survey) to simulate how altering discharge, groundwater input, channel wetted width, and shade prevents the temperatures of White Mountain, Arizona, stream reaches from exceeding the thermal tolerance of Apache Trout *Oncorhynchus apache*, both under existing conditions and under a climate change scenario. Simulations suggested increasing shade, either through streamside planting of specific numbers and species of plants or by other means, would be most effective and feasible for cooling the stream reaches we studied. Ponderosa pine *Pinus ponderosa* and Douglas fir *Pseudotsuga menziesii* provided the most shade followed in order by Engelman spruce *Picea engelmannii*, Bebb's willow *Salix bebbiana*, Arizona alder *Alnus oblongifolia*, and finally coyote willow *Salix exigua*. Vegetation survival depends on the appropriateness of site conditions at present and under climate change, and planting in buffer strips minimizes additional water removal from the watershed through evapotranspiration. Alternative shading options, including thick sedge growth, shade cloth, or felled woody vegetation, may be considered when environmental conditions do not support plantings. Increasing groundwater input can cool streams, but additional sources are scarce in the region. Decreasing the width-to-depth ratio would succeed best on reaches with widths greater than 2.0 m. Increasing discharge from upstream may lower water temperature on reaches with an initial discharge greater than 0.5 m^3/s. Existing models provide suggestions to cool stream reaches. Further development of accessible software packages that incorporate evaporation, fragmentation, and other projected climate change effects into their routines will provide additional tools to help manage climate change effects.

Warming and fragmentation of streams due to climate change will continue to change North American fish communities (Daufresne and Boet 2007; Williams et al. 2009; Isaak et al. 2011; Wenger et al. 2011; Roberts et al. 2013; Reynolds et al. 2015; Lynch et al. 2016). Past research has been typically used to predict broad-scale climate effects over watersheds or regions or to interpolate temperature and associated variables between monitored sites (Steel et al. 2016; Isaak et al. 2017). Increasingly, research is investigating the development of management tools for fisheries biologists to address effects on local scales (e.g., Dege et al. 2016; Paukert et al. 2016).

Managing water temperatures at a specific site requires knowledge of how the temperature of the surrounding environment affects the water temperature of a particular stream. Water temperatures fluctuate when energy moves in and out of a stream (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 point to a downstream point is a combination of net radiation, conduction, advection, convection, condensation, and evaporation (Brown 1983; Moore et al. 2005). The major sources of energy (heat) entering a stream that can be altered by fishery biologists are shortwave radiation from the sun and advection from incoming surface flows and groundwater (Rowe 1963; Johnson 2004).

The amount of shortwave (i.e., solar) radiation reaching the stream affects stream temperature the most (Brown and Krygier 1970; Brazier and Brown 1973; Hewlett and Fortson 1982; Johnson 2004) and can by influenced by the surrounding topography (Dubayah and Rich 1995; Rutherford et al. 1997) and riparian vegetation (Larson and Larson 1996; Kauffman et al. 1997). Riparian vegetation is most commonly managed to reduce stream temperatures (Kauffman et al. 1997), and canopy height, distance of vegetation from the streambank, and vegetation type all affect its shading potential (Larson and Larson 1996; Beschta 1997).

Volume and temperature of surface flows and groundwater also influence the amount of energy entering a stream (Ward 1985; Beschta et al. 1987; Evans and Petts 1997; Fritz et al. 2006). Surface flow can enter either from upstream or from tributaries and is similarly affected by solar radiation, water inputs, and other factors. Tributaries are often smaller than the main channel and sometimes have a warming effect (Ward 1985). Groundwater temperatures fluctuate less than temperatures of surface water; thus, large groundwater input can minimize daily changes in stream temperature (Evans and Petts 1997).

The speed at which water temperature changes with the input of energy from solar radiation, groundwater, and surface flows is influenced by the stream's discharge and cross-sectional channel shape. More energy is required to heat water in streams with higher discharge, and these streams also take longer to cool (Brown and Krygier 1970; Constantz et al. 1994; LeBlanc et al. 1997; Ruochuan et al. 1998). Furthermore, because most energy exchanges occur at the air–water interface, a stream with a large width-to-depth ratio will have a greater surface area of water in contact with the air than a stream with a small width-to-depth ratio and therefore will heat and cool faster (LeBlanc et al. 1997). Reducing stream width increases the water velocity, decreasing the retention time of the water. Fast-moving water spends less time in contact with potential heat influxes (Johnson 2004) and thus does not heat up as quickly.

Increasing air temperatures and drought associated with climate change can increase water temperatures because of the close relationship between air temperature and water temperature (Stefan and Preud'homme 1993; Mohseni and Stefan 1999) and can result in a smaller volume of water subjected to the influence of a warming climate.

Accounting for the effects of multiple energy inputs and losses can create a challenge for predicting the effects of specific techniques to manage stream temperatures. Fortunately, stream temperature models exist that can demonstrate how changes in riparian vegetation, stream discharge, groundwater input, and width-to-depth ratio alter the energy entering and leaving a stream, thus affecting stream temperature (LeBlanc et al. 1997; Rutherford et al. 1997; Chen et al. 1998; Blann et al. 2002; Whitledge et al. 2006). Modeling approaches have examined how shading (Cristea and Burges 2010; Johnson and Wilby 2015; Seixas et al. 2018), environmental water transfers (Elmore et al. 2016; Null et al. 2017), a combination of shading and stream narrowing (Justice et al. 2017), shading and groundwater inputs (Wawrzyniak et al. 2017), and a variety of heat inputs (Wondzell et al. 2019) affect streams and could be used to plan for climate change. Use and further development of mechanistic models might allow site-specific recommendations to be developed by those tasked with managing stream temperature changes on individual stream reaches. Additional information that could be generated includes comparison of shade provided by different vegetation species and their cooling effects; recommendations about the numbers of individual plants of particular species needed to cool streams by specific amounts; and how groundwater inputs, stream narrowing, shading, and other factors all interact at a particular site to affect stream temperature.

Here we show how one such model, the Stream Segment Temperature (SSTEMP) model (Bartholow 2000; U.S. Geological Survey, Fort Collins, Colorado), can be used to provide site-specific information to manage conditions that affect stream temperatures under climate change and other scenarios. Furthermore, we discuss how, with further research, models could be expanded to include other factors associated with climate change besides temperature and how simple predictive models could be combined with more complex models to provide important tools for site-specific management of stream temperatures. We applied this model to streams containing Apache Trout *Oncorhynchus apache* in the White Mountains of Arizona.

Salmonids on the far southern end of their range, such as Apache Trout, are among the most susceptible fishes to water temperature increases and other climate change effects. Other factors affecting Apache Trout populations include increased turbidity, loss of spawning and rearing habitats, alterations in stream productivity and food supply, and introductions of nonnative trout species (USFWS 2009). Apache Trout are endemic to the headwaters of the White, Black, and Little Colorado River drainages in eastern Arizona's White Mountains. Once abundant, the Apache Trout is now listed as threatened under the U.S. Endangered Species Act (Robinson et al. 2004; USFWS 2009) and climate change threatens the species' recovery.

Increasing stream temperature affects fish survival, growth, and reproduction (Hokanson et al. 1973; Xu et al. 2010; Whitney et al. 2016), and the upper thermal tolerances of several segments of Apache Trout life history are well known. Adult Apache Trout can survive short increases in water temperatures up to 28–31°C (Lee and Rinne 1980; Recsetar et al. 2012), but feeding stops at around 23°C (Lee and Rinne 1980). Apache Trout fry experienced approximately 40% mortality when water temperature fluctuated by ±6°C around a 22°C midpoint in a 30-d experiment (Recsetar et al. 2014), corresponding to a 30-d LT50 (temperature lethal to 50% of test fish) of 23°C. An understanding of how various physical factors influence temperatures of streams inhabited by Apache Trout and maintaining the temperatures below Apache Trout upper thermal tolerances are important for the species' continued survival.

Our specific objectives were to (1) quantify the degree to which existing environmental conditions, such as discharge, groundwater input, width-to-depth ratio, and shade, affect water temperatures of select White Mountain streams containing Apache Trout; (2) evaluate the effect of manipulating these conditions on stream temperatures; and (3) use this information to provide a suite of management options to cool streams under varying scenarios of increasing temperatures, focusing on the time of greatest thermal stress in the streams (May and June). We addressed three different scenarios:

- • *Cool streams by 1°C*: how to cool streams by 1°C when water temperatures are at their annual warmest. Lowering the maximum downstream temperature of each reach by the same amount (1°C) allowed us to compare how differences in environmental characteristics affect stream temperatures among reaches.
- • *Cool streams below the Apache Trout LT50 (existing conditions)*: how to keep all of the stream reach 0.1°C below the Apache Trout 30-d LT50 when stream reach temperatures are at their annual warmest.
- • *Cool streams below the Apache Trout LT50 (maximum climate change)*: how to cool stream reaches below the Apache Trout 30-d LT50 assuming that the average warmest air temperatures increase by 6°C, a maximum temperature increase predicted under climate change (Karl et al. 2009). This represents a "worst case under high emission" scenario; therefore, if stream temperatures can be effectively lowered to meet this temperature increase, they can be lowered to meet less drastic increases.

Methods

Study sites

We modeled effects of water temperature management scenarios on reaches (segments) of streams located in the White Mountains of Arizona. The White Mountains resulted from vulcanism during the middle Tertiary Period and were shaped by glaciation during the Quaternary Period (Merrill and Pewe 1977). The area is part of the Arizona/New Mexico Mountains Ecoregion, and historically land cover changes have been due to the logging of its boreal forests and due to forest fires (Ruhlman et al. 2012). Ungulate grazing and off-road vehicles are common near riparian corridors in the area (Nichols et al. 2017). In late May 2011, the Wallow Fire—the largest wildfire in Arizona history—swept through the Apache–Sitgreaves National Forest and burned 217,741 ha, so a necessary additional objective was to select study reaches with a minimum of fire damage.

Stream reaches studied were located on Conklin Creek, Hayground Creek, the West Fork Black River, and the West Fork Little Colorado River, all located within the Apache–Sitgreaves National Forest in the White Mountains of east-central Arizona (Figure 1). Conklin Creek flowed through Middle Miocene to Oligocene volcanic rocks, including basalt, andesite, dacite, and rhyolite (Arizona Geological Survey 2019). It had a watershed area of 76,127,300 m^2, 21% of which was forested. Riparian trees included Douglas fir *Pseudotsuga menziesii*, ponderosa pine *Pinus ponderosa*, Engelmann spruce *Picea engelmannii*, Gambel oak *Quercus gambelii*, Arizona alder *Alnus oblongifolia*, and coyote willow *Salix exigua*. The Wallow Fire of 2011 inflicted significant damage on the riparian area of Conklin Creek. We sampled a 3,245-m stream reach with an upstream point (33°40′16.4″N, 109°25′29.5″W; elevation = 2,297 m above mean sea level [amsl]) downstream of a large incoming tributary and a downstream point (33°40′55.0″N, 109°26′53.7″W; elevation = 2,208 m amsl) immediately upstream of a fish barrier.

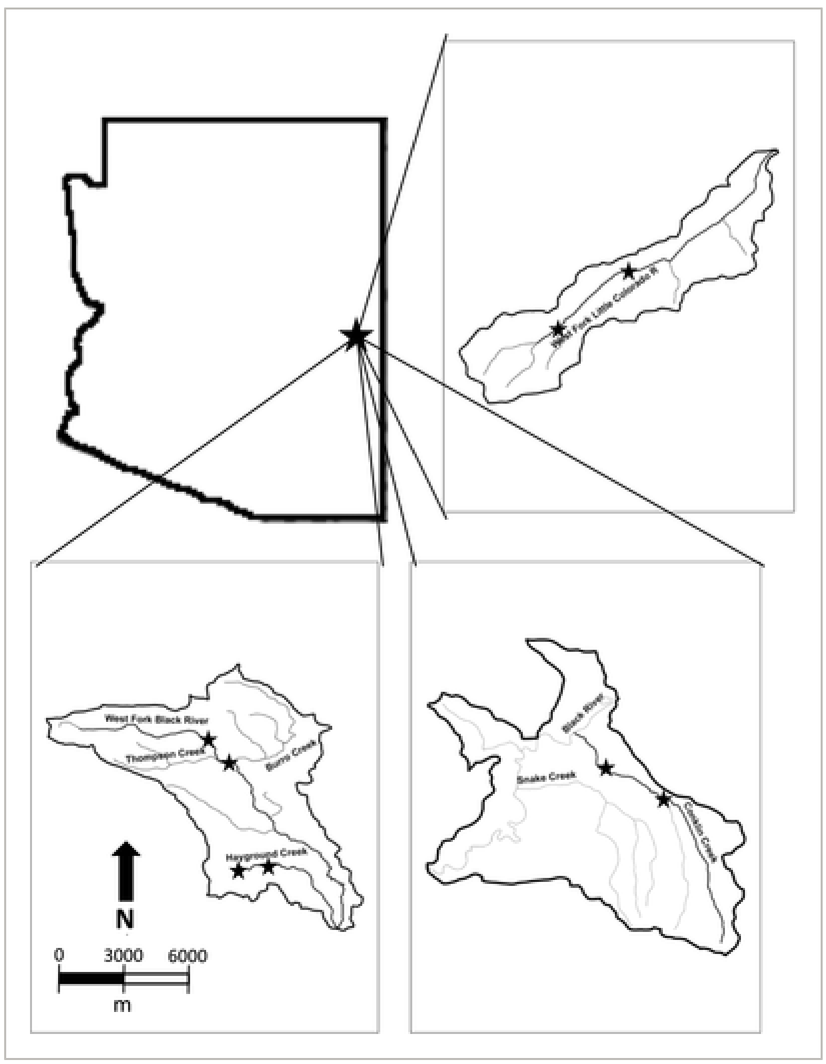

Figure 1

Open in figure viewer | PowerPoint

Locations of the West Fork Little Colorado River, West Fork Black River, Hayground Creek, and

Conklin Creek in Arizona. Upstream and downstream points of each studied reach are marked with stars. The West Fork Black River and Conklin Creek are tributaries of the Black River; Hayground Creek is a tributary of the West Fork Black River; and the West Fork Little Colorado River is a tributary of the Little Colorado River. Scale and direction are the same for all maps.

Hayground Creek also flowed through Middle Miocene to Oligocene volcanic rocks (Arizona Geological Survey 2019). It was the 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 2,000-m study reach flowed through an open meadow with only a few scattered Engelmann spruce. Fire damage to riparian vegetation within the segment was minimal. The upstream point of the study reach was near the headwaters, where the creek transitioned from shallow standing water to flowing water (33°50′16.4″N, 109°28′20.3″W; elevation = 2,721 m amsl), and the downstream point (33°50′22.4″N, 109°27′27.9″W; elevation = 2,690 m amsl) was directly upstream of a road crossing that caused the stream to pool.

The West Fork Black River flowed through Pliocene to Middle Miocene volcanic rocks consisting of rhyolite, andesite deposited as lava flows, and related rocks associated with basaltic rocks (Arizona Geological Survey 2019). It had a watershed area of 87,201,200 m^2, 31% of which was forested. The 2,215-m study reach flowed through an open meadow with scattered Engelmann spruce at the upstream point. Effects of the Wallow Fire on the riparian vegetation within the stream reach were minimal. The upstream point (33°53′44.0″N, 109°28′54.4″W; elevation = 2,717 m amsl) of the reach was just downstream of a densely vegetated canyon, where the stream flowed into a meadow. The downstream point (33°53′17.0″N, 109°28′19.8″W; elevation = 2,690 m amsl) was upstream of an incoming tributary.

The West Fork Little Colorado River flowed through Holocene to Middle Pliocene basaltic rocks with small amounts of andesite, dacite, and rhyolite (Arizona Geological Survey 2019). Riparian trees along the study reach were dominated by Engelmann spruce. The West Fork Little Colorado River reach was the largest of the four stream segments; it was the widest and longest segment with the highest level of discharge. Its watershed area was 32,912,260 m^2, 89% of which was forested. This stream was unaffected by the Wallow Fire. The upstream point (33°56′15.4″N, 109°32′19.2″W; elevation = 2,889 m amsl) of the 5,056-m study reach was placed downstream of a major incoming tributary. 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 (33°57′32.6″N, 109°30′38.5″W; elevation = 2,784 m amsl) was placed upstream of a series of beaver ponds.

Each reach (1) contained Apache Trout, (2) was delineated by how far water traveled uninterrupted for a 24-h period downstream (as confirmed by dye movement tests; length varied by velocity of flow), and (3) contained no major tributaries or large pools that could have led to a decrease in accuracy in the modeling procedure. Three additional nearby streams (Canyon, Christopher, and Tonto creeks) were not included in modeling simulations but were surveyed to provide additional tree-specific shade data to improve simulations.

We used ArcMap (ESRI, Redlands, California) and TOPO (National Geographic, Margate, Florida) to establish the area (m^2) of each watershed for each stream reach. We obtained input data for modeling in mid- to late May 2011 and 2012, during the 2-month low-discharge period leading up to monsoon storms—a time in which Apache Trout distribution is most limited by high temperatures. Late-May discharge in 2011 and 2012 varied from 0.002 to 0.677 m^3/s and was among the lowest 10% ever recorded in White Mountain streams (USGS 2013). Temperature of the water flowing into each stream reach varied from 7°C to 11°C (Table 1).

Table 1. Initial hydrological conditions (SE in parentheses) for study reaches in four Arizona streams. Presented values include the average upstream water temperature obtained during the May sampling period, the estimated upstream water temperature from mean maximum air temperature conditions over a 12-year period, and the estimated average upstream water temperature when a 6°C increase in maximum annual air temperature due to climate change is applied. Rate of change (°C/m) is the measured rate of change in water temperature from the upstream point to the downstream point of the segment. Inflow and outflow discharge of each stream reach was measured during the May sampling period at the top and bottom of the reach and at 100-m transects throughout the reach for mean discharge. Width *a* and *b* terms were calculated from the log relationship between channel wetted width and discharge. Groundwater (GW) input was calculated based on the change in surface discharge

Stream	Reach length (m)	Mean reach width (m)	Width *a* term	Width *b* term	Water temperature					
					Sampled upstream average (°C)	Estimated upstream under maximum annual air temperature conditions (°C)	Estimated upstream under 6°C climate change scenario (°C)	Rate of change (°C/m)	Inflow (m^3/s)	O (
Hayground Creek	2,000	0.982 (0.147)	1.071	0.015	8.637	16.035	23.433	0.0016	0.005	(
Conklin Creek	3,245	2.1 (0.192)	2.493	0.052	11.202	14.770	18.338	0.0014	0.013	(
West Fork Black River	2,215	2.73 (0.195)	3.894	0.561	9.883	13.601	17.319	0.00045	0.677	(
West Fork	5,056	3.566	3.781	0.098	6.575	10.233	13.891	0.00039	0.385	(

Little Colorado River	(0.137)

Description of the temperature model

We chose SSTEMP (Bartholow 2000) to model stream cooling methods because (1) it was developed in 1984 and has since been tested and updated numerous times (Bartholow 2002); (2) it is deterministic, meaning that it is well suited for analyzing impact scenarios due to anthropogenic effects and the effects of manipulating model variables (e.g., shade, channel width, and discharge), but it is simple enough for examining our study streams (i.e., small reaches with few or no tributaries); (3) various studies have used it in field applications, with accurate results (Yoshida et al. 2004; Harper-Smith 2008; Callahan et al. 2015); (4) components of the model have been well-validated (Theurer et al. 1984; Bartholow 1989; Boyd and Kasper 2004); (5) it incorporates all of the factors (described above) that most affect the temperature of Apache Trout streams; and (6) there were means by which to ground-truth and calibrate the model to maximize its accuracy in our study streams.

The SSTEMP model uses a combination of hydrological, geometric, and meteorological data collected on site and model default values to estimate how characteristics of the stream reach affect water temperature (Bartholow 2000). The upstream temperature of water flowing into the reach is entered, and the model uses data input on the stream characteristics to predict the temperature at the lower end of the reach. The SSTEMP model is based on a 24-h time frame and predicts daily average, minimum, and maximum downstream temperatures; thus, it assumes that the upstream water temperature and meteorological data inputs to the model also include daily averages and maximums.

Model inputs

We used data from on-site surveys when possible to provide input to the model. Methods for obtaining data for the model are summarized below. A complete list of model inputs and a description of how these variables were obtained are provided by Bartholow (2000) and Price (2013).

Hydrology inputs include the temperature (°C) of water entering the reach from upstream (inflow), water discharge (m^3/s) at the beginning and end of the reach, and groundwater accretion temperature (°C). We measured inflow temperature with a HOBO pendant temperature/light data logger (Onset Computer, Cape Cod, Massachusetts) that was secured midstream at the upstream end of the reach and was completely submerged and not in contact with substrate or any other heat sink or source (e.g., rocks or logs; Oregon Plan for Salmon and Watersheds 1999). Loggers recorded water temperature every 15 min for 24 h. From this, we calculated the daily average water temperature at the upstream point and input the value into the model. Upstream temperature values that were used as inputs to the model varied by the scenario tested and are discussed below. Discharge was calculated as the product of wetted channel cross section (m^2) and water velocity, which was measured using a U.S. Geological Survey Model 6205 pygmy current meter and an AquaCalc Pro open channel flow computer (Rickly Hydrological, Columbus, Ohio) at the reach's upstream and downstream points. Mean annual air temperature (°C) can be an accurate substitution for groundwater accretion temperature when actual groundwater temperature cannot be measured (Bartholow 2000). Therefore, we used mean annual air temperature from 2011, which was obtained from the National Oceanic and Atmospheric Administration's National Weather Service station in Greer, Arizona, to represent accretion temperature.

Required geometry data inputs to the SSTEMP model include reach length, latitude, upstream and downstream elevations, a and b terms of width from a width–flow relationship (Leopold and Maddock 1953), and Manning's n (described below; Bartholow 2000). We used an Opti-Logic 1000LH laser rangefinder/hypsometer (Opti-Logic, Tullahoma, Tennessee) to calculate reach length and a Garmin GPSMAP 60CSx (Garmin International, Olathe, Kansas) to calculate latitude and upstream and downstream elevations. In SSTEMP, width is a function of discharge in the form of $W = aQ^b$, where W = width (m) of the stream, Q = mean discharge (m^3/s) of the stream, and a and b are empirically derived coefficients (Leopold and Maddock 1953). To calculate the values a and b for each reach, we 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 reach and ending at the downstream point. We took the natural logarithm of both stream width and discharge and performed a standard linear regression with discharge as the independent variable. The b term was the slope of the regression, and we 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). We used precalculated Manning's n-values from Chow (1959) 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 data required 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. We used a four-channel HOBO micro-station and a set of smart sensors (Onset Computer) to measure most of the meteorological data. We set the weather station near each stream reach but outside of the riparian vegetation. The station collected data every 15 min for one 24-h period—the same 24-h period in which the instream data loggers recorded water temperature. We used a HOBO 12-bit temperature RH smart sensor to measure air temperature (°C) and relative humidity (%), a HOBO wind speed smart sensor to measure wind speed (m/s), and a HOBO silicon pyranometer smart sensor to measure solar radiation. The measure for possible sun was obtained from a MesoWest weather station in Flagstaff, Arizona. Because we measured ground-level solar radiation, we did not use the 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, we entered mean annual air temperature obtained from a weather station in Greer, Arizona (Western Regional Climate Center 2012), into the model for ground temperature (Bartholow 2000). The thermal gradient, a unitless term, determines the rate at which heat is lost or gained from the streambed to the water. Since small changes to this parameter do not affect downstream water temperature, we used the model's default value of 1.65 (Bartholow 2002).

We calibrated the model using air temperature and associated meteorological data collected on site by our portable weather station. However, evaluation of factors for cooling streams would be improved using temperature data that were representative of the area's climate—not simply a snapshot of 1–2 years. Therefore, for use in simulations, we obtained air temperature and associated data extending over a 12-year period from a nearby weather station in Greer, Arizona (MesoWest, University of Utah, unpublished data). We averaged daily and maximum air temperatures, relative humidity, and solar radiation from the hottest day of the year for the period 2001–2012 to account for annual variability. We define this average of the 2001–2012 maximum air temperatures as “existing climate conditions” to differentiate it from the 6°C climate change scenario that was also tested. The Greer station was one of the closest weather stations to streams containing Apache Trout, as it was located 43, 24, 18, and 11 km from the start of the Conklin Creek, Hayground Creek, West Fork Black River, and West Fork Little Colorado River reaches, respectively. However, it was 244–305 m lower in elevation than some of the highest Apache Trout streams. Using air temperature data from Greer ensured that we would be planning for worst-case, warmest temperature scenarios for the higher-elevation Apache Trout streams.

We also had to adjust the temperature of the water entering the reach under different scenarios because air temperature would be expected to increase in a warming climate. The relationship between air temperature and stream temperature is not always linear (Mohseni and Stefan 1999). However, because our main objective was to evaluate success and feasibility of various management activities for cooling streams under the same temperature scenario, we assumed that a simplified relationship between stream temperature and air temperature, which was based on 43 river and stream sites in 13 different countries (Morrill et al. 2005), would be adequate. This relationship stated that for every 1°C increase in air temperature, stream temperature increased by 0.7°C. Therefore, to estimate the average upstream water temperature in each simulation (Table 1) we 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. This increase in air temperature was multiplied by 0.7, and the resulting value represented the average upstream temperature increase for each scenario. This temperature was added to the average upstream temperature we measured in the field during mid-May. After entering these data into the model, we obtained estimates of downstream reach temperatures under climate warming scenarios without any adjustments of stream characteristics.

The shading parameters we measured included reach azimuth; topographic altitude; and vegetation height, crown, offset, and density. We estimated reach azimuth—the general orientation of the stream reach with respect to due south—by using a compass. We measured the topographic altitude (line-of-sight angle to the horizon from the middle of the stream) and vegetation density, height, offset, and crown at each transect to obtain data every 100 m, starting at the upstream point of the reach 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 (density quantity) and percentage of light filtered by leaves and trunks (density quality). Density quantity was calculated as the percentage of left or right banks across all transects at which trees were within 35 m of the stream—the maximum distance from which vegetation can effectively shade a stream (Bartholow 2000). Where trees were closer than 35 m, we used an Extech Model 401025 lux meter (Extech Instruments, Nashua, New Hampshire) to calculate vegetation density quality. At each transect, on both sides of the stream, we measured lux outside of the riparian area in full sunlight and then under the tree closest to the stream to estimate the percentage of light filtered by the individual tree. We averaged the quality measurements of each transect for the east side and the west side. We then 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.

We measured vegetation offset (m) at the location of each transect as the perpendicular distance from the wetted streambank to the closest tree. We measured the height and crown (m) of the closest tree. When we measured vegetation height, we included bank height as well. Vegetation crown was measured as the width from tip of branch to tip of branch at its widest point. We used the laser hypsometer to measure topographic altitude and vegetation offset, height, and crown (Opti-Logic 2004). We 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 as input into the model.

We recorded the shade tree species at each transect and calculated relative abundance of each species along the stream reaches. We calculated average values of vegetation density quality (amount of light filtered), crown, and height for the six most abundant shade species. To provide additional data on shade characteristics of the common riparian plant species of the area, we collected data on plants in three nearby streams, which constituted an additional 9.7 km of stream, totaling 90 transects.

Model calibration

We entered all model inputs into SSTEMP and predicted the daily average, maximum, and minimum water temperatures at the downstream end of the reach. To assess how accurately the model predicted downstream temperature, we placed a HOBO pendant temperature/light logger at the downstream point of each reach. We used information obtained from the downstream temperature sensor to calibrate the model. Because we were interested in the daily maximum downstream temperature, we 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). We performed this calibration separately for each stream reach, and each new wind speed was used for the remaining model simulations.

Testing of temperature scenarios

To test each of the three scenarios, we first estimated initial stream conditions present under each scenario and the associated downstream temperature. We then modified stream conditions (i.e., average discharge, stream width, and percent shade) to evaluate which, if any, manipulation could bring the downstream water temperature of each reach to the desired temperature goal in each scenario. We calculated the rate of longitudinal temperature change for each stream reach by dividing the difference in daily average stream temperature from the upstream point to the downstream point by the distance between the two points.

Effects of in-channel modifications on stream temperatures were tested in different ways. Effects of altering discharge were tested by adjusting the inflow and outflow discharges by the same amount within the model.

Groundwater input and lateral inflow input are assumed if discharge increases from the upstream point to the downstream point (Bartholow 2002). Because we made measurements during base flows several months after any major precipitation and because we avoided all major tributaries, any increase in stream discharge from the upstream point to the downstream point was due primarily to groundwater inflow. Therefore, we increased reach outflow—while keeping reach inflow the same—to estimate the increase in groundwater (m^3/s) needed to reach temperature goals under each scenario. We tested the effects of a decreasing stream width-to-depth ratio on water temperatures by setting the *b* term of the width to zero within the model and by varying the *a* term of the width (Bartholow 2002).

We 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.). For each of the six most common shade species found along streams in the White Mountains, we then estimated the additional number of trees per meter of stream reach that would be needed to achieve the required additional shade through vegetation plantings. We used average values for crown, height, and density quality for each species, as calculated from field measurements, to add trees one by one to the model. The majority of the deciduous tree values (80%) came from trees that were measured in the Tonto National Forest during June and therefore were fully leafed-out at the time of sampling. All trees added in the simulations had an offset of 0 m.

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

Statistical analysis

We deemed a cooling method "successful" if by using the method (e.g., increasing shade, increasing discharge, or narrowing stream width) we could lower reach temperature in the model to successfully attain the scenario temperature goal. We compared the effectiveness of the methods by summing the number of times each method successfully attained a temperature goal for each stream reach × scenario combination. We then performed a chi-square test to evaluate the null hypothesis ($\alpha = 0.05$) that the number of scenario × stream combinations where reach temperature was lowered to the desired goal did not differ among cooling methods. For shading data, we performed a one-way ANOVA and Tukey–Kramer post hoc analysis ($\alpha = 0.05$) to evaluate whether there were significant differences among species in the number of trees per 100 m required to reach the desired level of cooling.

Results

Model Inputs

Hydrological data were successfully obtained to characterize the streams (Table 1). The Hayground Creek reach had the highest estimate of annual average upstream temperature, followed by the Conklin Creek, West Fork Black River, and West Fork Little Colorado River reaches. The West Fork Little Colorado River reach 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 reaches. The West Fork Little Colorado River reach had the greatest discharge at the lower end, followed by the West Fork Black River, Conklin Creek, and Hayground Creek reaches. The West Fork Black and West Fork Little Colorado River reaches experienced an influx of groundwater, whereas the Conklin and Hayground Creek reaches lost discharge from the upstream point to the downstream point. The West Fork Little Colorado River reach was widest, followed by the West Fork Black River, Conklin Creek, and Hayground Creek reaches. The natural logarithm relationship between stream discharge and channel wetted width of the stream reach at the 100-m-spaced transects within the reach (Figure S1 available in the online version of this article) was successfully used to derive the a and b terms for the width equation.

On-site meteorological data obtained during the May sampling period (Table S1 available in the online version of this article) were within the range of values typically measured by nearby official weather stations. All data were successfully used to calibrate the model to stream conditions.

The West Fork Little Colorado River and Conklin Creek reaches had the highest densities of riparian vegetation providing the highest percentage of total shade (41% and 49%, respectively), followed by West Fork Black River (5%) and Hayground Creek (2%; Table S2). The Conklin Creek reach had the most diverse riparian tree and shrub community, consisting of 47% Douglas fir, 32% ponderosa pine, 5% Arizona alder, 3% Engelmann spruce, 1.5% coyote willow, 10% other species, and 1.5% unvegetated. Hayground Creek, West Fork Black River, and West Fork Little Colorado River were much less diverse and more open, consisting of 30% Engelmann spruce, 10% ponderosa pine, and 60% unvegetated; 9% Engelmann spruce and 91% unvegetated; and 88% Engelmann spruce, 8% coyote willow, 2% Bebb's willow *Salix bebbiana*, and 2% unvegetated, respectively.

Of the six most common shade trees and shrubs along the stream reaches, 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 2). Ponderosa pine was the largest tree with the greatest mean height and mean crown.

Table 2. Vegetation characteristics (means with SE in parentheses) for the six most common riparian vegetation species along the streams sampled

Species	Mean height (m)	Mean crown (m)	Mean density quality (%)
Douglas fir	24.362 (1.446)	7.818 (0.371)	82.40 (0.034)
Engelmann spruce	17.572 (0.594)	7.179 (0.233)	78.788 (0.032)
Ponderosa pine	24.450 (1.103)	8.833 (0.293)	76.973 (0.033)

Arizona alder	7.673 (0.570)	5.583 (0.332)	77.640 (0.034)
Bebb's willow	6.353 (0.765)	5.808 (0.473)	78.131 (0.039)
Coyote willow	3.027 (0.317)	4.106 (0.559)	67.615 (0.089)

Model Calibration

The difference between the actual maximum downstream temperature measurements in the field during mid-May and the values predicted by the model averaged 0.28°C (SE = 0.162) over all stream reaches and never exceeded 1°C for any one reach (Table 3). Wind speed was altered by an average of 0.21 m/s (SE = 0.089) to calibrate predicted temperature to match actual temperature.

Table 3. Maximum downstream temperature predicted by the Stream Segment Temperature model, maximum downstream temperature sampled in the field, the estimate of maximum downstream temperature when maximum annual air temperature was modeled, and the estimated maximum downstream temperature when a 6°C increase in maximum annual air temperature due to climate change was modeled

Stream	Predicted maximum downstream temperature (°C)	Sampled maximum downstream temperature (°C)	Estimated downstream maximum temperature with maximum annual air temperature (°C)	Estimated downstream maximum temperature under a 6°C 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

Baseline Conditions for Modeled Temperature Scenarios

The Conklin Creek reach had the highest estimate of annual maximum downstream temperature, followed by the Hayground Creek, West Fork Black River, and West Fork Little Colorado River reaches (Table 3). The estimate of annual maximum downstream temperature was 7.46°C above the Apache Trout LT50 for the Conklin Creek reach and 5.62°C above the LT50 for the Hayground Creek reach. The estimates of annual maximum downstream temperature on the West Fork Black and West Fork Little Colorado River reaches did not exceed the LT50 of Apache Trout. Under a 6°C 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 reaches were 10.24, 8.22, 2.49, and 1.33°C, respectively, above the LT50 of Apache Trout (Table 3).

Manipulating Environmental Variables to Cool Streams under Each Scenario

Stream discharge

The West Fork Black River reach required the greatest increase in incoming surface flow from upstream to lower the modeled estimate of annual maximum downstream temperature by 1°C and under existing climate conditions, followed by the West Fork Little Colorado River, Hayground Creek, and Conklin Creek reaches (Table 4). We could not decrease the annual maximum downstream temperature under a 6°C climate change scenario below the LT50 for Apache Trout on the Hayground Creek and West Fork Black River reaches by increasing incoming stream discharge within the bounds of the model. We confirmed that higher-discharge streams gained less heat per distance by regressing the rate of water temperature change over distance with discharge (R^2 = 0.993, $F_{1,3}$ = 294, P = 0.003).

Table 4. Increase in surface flow, increase in groundwater input, or decrease in stream width (percent change from initial is shown in parentheses) needed to cool stream reaches by 1°C, below the Apache Trout LT50, or below the Apache Trout LT50 after a maximum climate change scenario (6°C increase in mean air temperature). Temperature goals that were unobtainable by increasing discharge are indicated by an en dash (–). Maximum downstream temperature of the stream reach not exceeding the Apache Trout LT50 under baseline conditions is indicated by “NA.”

Stream	1°C	Below LT50	Below LT50 (maximum climate change scenario)
Increase in surface flow from upstream (m^3/s)			
Hayground Creek	0.042 (840)	0.472 (9,440)	–
Conklin Creek	0.038 (292)	0.626 (4,815)	5.071 (39,008)
West Fork Black River	0.303 (45)	NA	–
West Fork Little Colorado River	0.201 (52)	NA	0.362 (94)

Increase in groundwater input (m^3/s)			
Hayground Creek	0.019 (950)	0.088 (4,400)	0.134 (6,700)
Conklin Creek	0.020 (286)	0.143 (2,043)	0.213 (3,043)
West Fork Black River	0.138 (16)	NA	0.313 (37)
West Fork Little Colorado River	0.056 (12)	NA	0.068 (14)
Decrease in stream width (m)			
Hayground Creek	–	–	–
Conklin Creek	1.395 (66)	–	–
West Fork Black River	0.602 (22)	NA	2.303 (84)
West Fork Little Colorado River	0.559 (16)	NA	0.769 (22)

Groundwater input

We successfully lowered the modeled estimates of annual maximum downstream water temperature of each stream reach by 1°C and below the LT50 of Apache Trout under existing climate conditions and under a 6°C climate change scenario by increasing groundwater input. Relative to the West Fork Black and West Fork Little Colorado River reaches, the Hayground and Conklin Creek reaches required a smaller increase in groundwater input (m^3/s) to lower stream temperatures by the same amount (Table 4). However, the Hayground and Conklin Creek reaches required a larger *percent* increase in groundwater input than the West Fork Black and West Fork Little Colorado River reaches to lower stream temperatures by the same amount.

Stream width

The Conklin Creek reach required the largest decrease in stream width to lower the modeled estimate of annual maximum downstream temperature by 1°C, followed by the West Fork Black and West Fork Little Colorado River reaches (Table 4). We could not lower the estimate of annual maximum downstream temperature of the Hayground Creek reach even by 1°C via decreasing stream width. Furthermore, we could not decrease the annual maximum downstream temperature below the LT50 for the reaches on Hayground and Conklin creeks by decreasing stream width under any of the scenarios.

Shade

Modeling indicated that by increasing shade, we could lower annual maximum downstream temperature of each stream reach to meet temperature goals under all three scenarios (Table 5). Depending on the stream reach and goals, the necessary increases in shaded areas would range from 191 to 3,535 m^2.

Table 5. Percent and area increase in total shade needed to cool stream reaches by 1°C, below the Apache

Trout LT50, or below the Apache Trout LT50 after a maximum climate change scenario (6°C increase in mean air temperature). "NA" indicates that maximum downstream temperature of the stream reach did not exceed the Apache Trout LT50 under scenario conditions

Stream	1°C		Below LT50		Below LT50 (maximum climate change scenario)	
	Percent increase	Area increase (m^2)	Percent increase	Area increase (m^2)	Percent increase	Area increase (m^2)
Hayground Creek	9.703	191	50.345	989	76.607	1,505
Conklin Creek	5.404	368	36.325	2,475	51.871	3,535
West Fork Black River	10.230	619	NA	NA	27.658	1,672
West Fork Little Colorado River	6.181	1,114	NA	NA	9.080	1,637

Among the six most common tree or shrub species, ponderosa pine provided the most shade per tree, while coyote willow provided the least. This was based on the number of trees per 100 m of stream that was needed to lower the estimates of maximum downstream temperature of each stream reach by 1°C. When tree species were compared within each of the stream reaches and not across all stream reaches, the ability of each species to lower annual maximum downstream temperature by 1°C differed (one-way ANOVA with Tukey–Kramer post hoc analysis: $F = 39.0076$, df = 5, $P < 0.0001$). In order, from the most shade provided to the least (species without a letter in common provide significantly different shade) are Douglas fir (w), ponderosa pine (w), Engelmann spruce (wx), Bebb's willow (xy), Arizona alder (y), and coyote willow (z). An average of 57% (SE = 4.51) more Arizona alders and Bebb's willows than conifers was required to lower the maximum downstream temperatures by the same amount, and an average of 172% (SE = 13.40) more coyote willows than conifers was required to lower maximum downstream temperatures by the same amount.

The Hayground Creek reach required the highest number of trees per 100 m of stream to lower the estimate of annual maximum downstream temperature by 1°C. This was followed by the West Fork Black River reach, the Conklin Creek reach, and finally the West Fork Little Colorado River reach (Table 6). We could cool the estimated maximum downstream temperature of each stream reach below the Apache Trout LT50 under existing climate conditions by adding riparian vegetation (Table 6). However, if we only used coyote willow we could not lower the estimated maximum downstream temperature of the Conklin Creek reach below the LT50. The estimated annual maximum downstream temperature was cooled from 30.36°C to 24.38°C, which was still 1.48°C above the LT50. After accounting for a 6°C increase in air temperature, planting any species of riparian vegetation was still successful at lowering the West Fork Black and West Fork Little Colorado River reaches below the LT50 (Table 6). On the Hayground Creek reach, ponderosa pine, Douglas fir, and Engelmann spruce could each be used independently to lower the estimated maximum downstream temperature 0.1°C below the LT50 after a 6°C increase in air temperature. However, entering the maximum number of Bebb's willows into the model resulted in an estimated annual maximum downstream temperature of 22.89°C, which is only 0.01°C below the LT50 for Apache Trout. Entering the maximum number of Arizona alders into the model resulted in an annual maximum downstream temperature of 22.96°C, which was 0.06°C above the LT50, and entering the maximum number of coyote willows into the model resulted in an estimate of annual maximum downstream temperature of 24.13°C, which was 1.23°C above the LT50. On the Conklin Creek reach under a 6°C 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 LT50. Entering the maximum number of trees into the model resulted in an average decrease in the estimated annual maximum downstream temperature of 7.12°C for each vegetation species. On average, this was still 3.12°C above the LT50 of Apache Trout.

Table 6. Additional number of trees (per 100 m of stream) needed to cool stream reaches by 1°C, below the Apache Trout LT50, or below the Apache Trout LT50 after a maximum climate change scenario (6°C increase in mean air temperature). Temperature goals that were unobtainable by increasing shade are indicated by an en dash (–). “NA” indicates that the maximum downstream temperature of the stream reach did not exceed the Apache Trout LT50 under baseline conditions

Stream	Douglas fir	Engelmann spruce	Ponderosa pine	Arizona alder	Bebb's willow	Coyote willow
1°C						
Hayground Creek	7.9	8.9	7.0	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
Below LT50						
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	NA	NA	NA	NA	NA	NA

River						
Below LT50 (maximum climate change scenario)						
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

Effects on evapotranspiration due to adding riparian vegetation to shade streams would be negligible. The amount of riparian vegetation needed to cool the estimate of maximum downstream temperature of the Hayground Creek reach by 1°C, below the Apache Trout LT50 under existing climate conditions, and below the LT50 under a 6°C climate change scenario would decrease the stream depth by an average of 3.5% (SE = 0.737). The average percent decrease in stream depth for the Conklin Creek, West Fork Black River, and West Fork Little Colorado River reaches 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 reach under a 6°C climate change scenario if ponderosa pines were added to the model. Adding conifers to the stream reaches would, on average, lower the stream depth 52% (SE = 1.66) more than adding willows and alders.

Success of Cooling Methods

Within the confines of the model, the success of altering stream discharge, width, shade, and groundwater input to cool the streams differed (χ^2 = 12.798, df = 4, P = 0.0123). Increasing the total percentage of shade along the stream reach was successful for all 10 method × stream simulation combinations. Using riparian vegetation to achieve this level of shade was successful for 9 of 10 simulations. Increasing the amount of groundwater input within the reach was successful for all 10 simulations. Increasing the incoming discharge to the stream reach was successful for eight simulations, whereas decreasing stream width was successful for only five simulations. Despite success within the model, choosing the option that will work best for each stream reach is often dependent upon feasibility.

Discussion

Most previous climate research has examined impacts across large areas and is less focused on providing management tools for individual streams. Recently, stream temperature models have been used to examine water temperature restoration techniques needed for climate change on individual streams and stream basins (e.g., Justice et al. 2017; Wondzell et al. 2019). Existing stream temperature models, which are used to evaluate specific management scenarios to cool streams, usually in response to anthropogenic activities (e.g., urbanization, livestock, or logging), could help to devise management actions for cooling specific streams that will warm due to climate change. However, as discussed further below, if these models are to be completely effective they will need additional development to incorporate other elements of climate change that are not currently included. Furthermore, the first step in evaluating the use of temperature models is to decide whether temperature—and not some other factor, such as introduced predators and competitors, loss of spawning and rearing habitats, and alterations in stream productivity and food supply (USFWS 2009)—is the most limiting factor.

Our simulations show how stream temperature models such as SSTEMP can give specific suggestions on how increasing shade, groundwater input, and flow from upstream may be used to lower temperatures of stream reaches to acceptable levels for fish. However, what is theoretically possible may not be feasible. Comparing the different methods to reduce temperatures of all stream reaches by 1°C and current characteristics of each reach provides ways to assess the feasibility of the different methods. The Conklin and Hayground Creek reaches required 286–950% increases in groundwater or upstream surface inflow to lower the reach temperature by 1°C (Table 4). No surface water was being diverted, and few or no groundwater wells were in the headwaters of these systems. Therefore, flow increases would be unavailable from transfers from current water diversions (Elmore et al. 2016; Null et al. 2017). Cutting watershed forests to increase water yield in Conklin and Hayground Creek reaches, which we discuss more below, would also be unlikely to provide sufficient water to cool the streams. The Conklin and Hayground Creek watersheds were only 21% and 18% forested at the time of this study. We examined percent increases in water yield in a subset of 28 watersheds where forests were 100% removed (data from Bosch and Hewlett 1982). In only one was increased water yield higher than 286%. Decreasing stream width would not sufficiently lower Hayground or Conklin Creek reach temperatures. Reducing Hayground Creek reach temperature by 1°C was impossible no matter the amount of narrowing. Conklin Creek would have to be narrowed by 66% (Table 4), an amount presumably narrower than that at which the stream channel would eventually equilibrate. Increases in shading seemed to be the most feasible for cooling these reaches. The Conklin Creek reach had a riparian area that was heavily damaged by fire, and the Hayground Creek reach was minimally shaded (1.873% total shade). Shade increases of 5.4% and 9.7% (1.6–16.6 coyote willows/100 m) would be required to cool the Conklin and Hayground Creek reaches, respectively, by 1°C. Under the maximum 6°C climate change scenario, the Conklin Creek reach could not be cooled by streamside vegetation shading, but this strategy could work for the Hayground Creek reach.

More options were feasible for cooling the West Fork Black and West Fork Little Colorado rivers. Groundwater input would have to increase 12–16% and surface input would have to increase 45–52% to cool these reaches by 1°C. Again, these headwaters were isolated, so environmental water transfers would not provide the needed flow. The forested portion of the West Fork Black River watershed was low (31%), but the West Fork Little Colorado River watershed was 89% forested. Watershed cutting could provide flow to cool stream temperatures somewhat, but drastic clearing (100%) may be required for limited results. Narrowing both reaches to cool them by 1°C was feasible (Table 4) but not for the West Fork Black River under the maximum 6°C climate change scenario. Here again, for both reaches increased shading was the most feasible option. Riparian shading would have to be increased by 10.2% (3.7 coyote willows/100 m) and 6.1% (1.6 coyote willows/100 m) to cool the West Fork Black and West Fork Little Colorado River stream reaches, respectively, by 1°C. Under the maximum 6°C climate change scenario, stream temperatures could be lowered to adequate levels for Apache Trout by increasing shading 27.7% and 9.1% on the West Fork Black and West Fork Little Colorado rivers, respectively.

We agree with others studying a wide variety of streams in different geographic areas (e.g., Sugimoto et al. 1997; Whitledge et al. 2006; Harper-Smith 2008; Guoyuan et al. 2012; Wawrzyniak et al. 2017; Seixas et al. 2018; Wondzell et al. 2019) that increasing shade provided by riparian vegetation is one of the most effective, feasible means to keep streams from heating. Planted vegetation has a minimal effect on stream levels when applied sparingly as buffer strips and not watershed treatments. However, its degree of influence depends on stream location; stream width; groundwater input; and the width, composition, and density of the riparian buffer strip (Osborne and Kovacic 1993; Whitledge et al. 2006; Cristea and Burges 2010; Seixas et al. 2018). Furthermore, planting of vegetation provides additional benefits, such as buffering sediment and pollutant input, stabilizing banks, and providing food and nutrients to aquatic organisms. Active planting gives the manager more control over shading and can speed shading efforts. Plants can regrow through grazer (livestock and elk) exclusion alone, but this requires several decades for full recovery in some systems (Nussle et al. 2017) and specific shading requirements may not be met. Shade can also be provided by artificial techniques, such as shade cloth and felled vegetation (Kiffney et al. 2004; Matney 2004; Gothreaux and Green 2012), but these strategies are better as short-term solutions (e.g., forest fire recovery) because living vegetation provides the benefits listed above and typically is more esthetically pleasing. Increasing shade provided by vegetation promises to be an important tool for combating stream warming due to climate change.

Unlike most previous studies, we compared shade provided by specific tree species. Of the trees we studied, we found that conifers provided the greatest amount of shade per tree. However, conifers take 40 years on average to reach the height we measured in the field and modeled (USDA NRCS 2013). Alders (Featherstone 2012) and willows (Nellessen 2004) require only 10 years on average to reach the height we measured and modeled, so the desired level of shading is achieved four times quicker when planting willows and alders than when planting conifers. We compared species that were common in North American boreal forests. In warmer regions, obtaining shading and water use characteristics of local species would be useful.

We measured the shade characteristics of individual trees, which on average prevented 76.9% of sunlight from reaching the streams (density quality). Other planting techniques, such as layering vegetation types or providing shade from smaller plants (e.g., sedges), might block even more light from reaching the stream and could serve to further lower stream temperatures. Tests of various types of plantings to achieve more stream shading would be beneficial.

Factors besides shading potential are equally important when deciding which tree species to plant. Regional climate, stream gradient, elevation, soil, aspect, topography, water quantity and quality, and existing plant community—and how these characteristics will transform under a changing climate—will influence which plant species survive over time at a location (Oakley et al. 1985; Prentice et al. 1992; Kauffman et al. 1997; Richardson et al. 2007). Choosing resilient trees that can readily subsist under varying conditions, such as when the climate warms, would help to offset tree death when atmospheric temperatures increase or when precipitation levels change.

Other methods can cool streams but were less practical at our sites. The amount of surface flow entering a reach is influenced by various upstream factors, including precipitation, vegetation amounts, streambed composition, and groundwater and tributary 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), actions that increase groundwater recharge upstream of the affected reach may benefit these systems. Variation in the effectiveness of groundwater for cooling streams is due to the stream's position with respect to groundwater flow, geologic characteristics of the streambed (Brunke and Gonser 1997), and climatic variables (Winter 1999). In our simulations, stream reaches with the greatest amount of existing groundwater input required the smallest *percent* increase in groundwater input to cool them. Sources of additional discharge to southwestern U.S. stream reaches, either surface flow or groundwater, will be more difficult to locate as climate change progresses and as human populations increase. In other areas where water is already being diverted, environmental water transfers (Elmore et al. 2016; Null et al. 2017), in which payments are made to put water back into the stream, may be effective for increasing flow.

Cutting vegetation in the watershed to increase downstream flow has been extensively investigated. In a review of 39 studies, Hibbert (1965) reported that water yield could increase up to 4.5 mm/year with each percent reduction in forest cover; however, most catchments produced less than half this amount. Rowe (1963) recommended that vegetation removal treatments be focused on (1) areas with a water supply that is adequate to exceed evapotranspiration losses after treatment; (2) areas where the zone of saturation is within reach of the vegetation using the most water; and (3) areas where the soils above the water table are of sufficient extent and depth to permit a reduction in evapotranspiration if the vegetation is removed. However, Ellison et al. (2012) suggested that tree thinning does not always lead to increased water yield, and they cited evidence that the presence of forests increases the intensity of the hydrologic cycle, thus providing more precipitation to an area. Ultimately, they argued 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 increase 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. Cutting efforts focused on trees that do not shade the stream and on those that use a significant amount of water maximize the chances of this technique succeeding.

In our study streams, the narrowing of reaches was less feasible for cooling than streamside planting. Nevertheless, some have found that narrowing and deepening a stream (i.e., decreasing the width-to-depth ratio) could be combined with riparian revegetation efforts (Justice et al. 2017) and can occasionally have a greater effect on stream temperature than increasing riparian vegetation (Blann et al. 2002). 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. Furthermore, water velocity increases if a stream reach is narrowed, thus lessening the time water is in contact with potential heat influxes (Johnson 2004). We found that wide stream channels with the greatest discharge required the smallest decreases in width to lower downstream water temperatures. Management options to narrow a stream include eliminating the sources of bank erosion; fencing to protect streamside vegetation from grazing cattle and wildlife, which encourages growth of dense vegetation on exposed streambanks to trap sediments; actively decreasing the width-to-depth ratio by using large boulders, logs, and rootwads to stabilize banks and narrow the channel (Rosgen 1997); planting grasses and sedges along the streambanks to store sediments (Trimble 1997); or in extreme cases, reconstruction of the entire channel (Klein et al. 2007).

We did not study the 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 can help to prioritize where to make stream alterations. Bedrock-lined streams can exhibit higher temperature fluctuations than alluvial reaches (Johnson 2004). Smooth bedrock surfaces reflect solar radiation back into a shallow stream. 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). Furthermore, 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 may provide thermal refuges for coldwater species, yet the 100 pools and beaver ponds naturally occurring in two White Mountain streams did not provide thermal refuge because they were too well mixed and too shallow (Bonar and Petre 2015). Pools would have to be much larger and deeper than those currently found in the streams we studied to provide thermal refuge. When pools are large enough to stratify by temperature, they provide thermal refuge for various salmonid species (Nielsen et al. 1994; Elliott 2000; Tate et al. 2007). For a pool to stratify, a source of cold water must exist, usually from tributaries or the streambed (hyporheic or groundwater input; Nielsen et al. 1994). Proximity of a pool to a source of cold water can be more important than pool size (Matthews and Berg 1997). For a pool to remain stratified, mixing must be stopped or weakened (Nielsen et al. 1994). Nielsen et al. (1994) found that pools over 3 m deep stratified when surface flow decreased to 1 m^3/s and the temperature at the bottom of the pools averaged 3.5°C cooler than the temperature at the surface.

Potential sources of error existed in our simulations. We estimated annual maximum upstream and downstream temperatures for each reach based on annual maximum air temperatures averaged over a 12-year period. Furthermore, we 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 temperatures are accurate surrogates for these parameters, yet measuring them in the field would have been more exact. We used an air temperature–water temperature relationship developed from geographically diverse sites (Morrill et al. 2005), which yielded a general relationship of a median 0.7°C change in water temperature with every 1°C increase in air temperature. This relationship could be fine-tuned in future modeling efforts by estimating a nonlinear relationship between air and water temperatures on site. Our streams exhibited substantial variability in the discharge–width relationships used to estimate the a and b terms of the width equation. The discharge–width relationship becomes more variable at low flows (Bartholow 2002) and across stream cross sections, characteristic of our measurements, and may contribute to more variability in model estimates. Finally, we did not vary input temperature when we modeled the effects of different discharges on downstream temperature. When discharge is increased from upstream, it would likely enter the reach cooler because larger volumes could not warm as much upstream. Being able to estimate the change in water temperature entering the reach due to increased discharge upstream would further improve model predictions.

Certain assumptions about the stream reach and data input to the model were required by SSTEMP. The stream reaches we studied largely fit model assumptions in that they were thoroughly mixed and had no major tributaries and flow took 24 h to reach the downstream point from the upstream point. Small violations of the assumptions (e.g., West Fork Black and West Fork Little Colorado River reaches were too short to adhere to the 24-h rule) did not seem to greatly affect model predictions, which were generally quite accurate.

Stream temperature models themselves are not infallible. As is the case for any model, not all variables affecting temperature could be included in SSTEMP. Altering riparian vegetation can affect channel width, air temperature, relative humidity, and wind speed (Bartholow 1989), having unknown effects on stream temperatures. Increasing discharge within the stream may widen the channel, leading to a different maximum downstream water temperature than predicted. Groundwater input to the reach was considered independently in simulations; however, it would likely be accompanied by an increase in surface flow entering the reach from groundwater contributions upstream. Accounting for this increase in upstream surface flow would likely improve future model predictions. Factors that are unique to a stream, including climate, terrain, surrounding vegetation species, and soil type, can lead to error when judging the efficacy of a vegetated buffer strip for altering stream temperatures (Barton et al. 1985; Guoyuan et al. 2012). Despite possible sources of error and limitations, the 30-year history of the SSTEMP model and its use in many other field applications with accurate results demonstrate this model's usefulness for understanding how different stream cooling methods relate to one another under varying stream conditions.

Current stream temperature models could be further optimized for climate change scenarios. These models only examine means of altering stream temperature and do not examine other effects of climate change, such as fires and sedimentation, extensive evaporation, lowered water level, and associated stream fragmentation. Inclusion of these factors provides a rich area for future research. Perhaps future models could incorporate a prioritization routine to first identify the most likely limiting factor to a stream reach (e.g., stream warming, fragmentation, forest fires, and riparian community shifts) and then identify management alternatives to address limiting factors in addition to water temperature. Nevertheless, currently available stream temperature models provide a valuable starting point and can serve as a call to action for designing increasingly useful tools. Furthermore, these models may be suitable in their present state where water availability changes little and where the major effect of climate change is exerted on stream temperature.

In summary, our modeling suggested that increased shading in Arizona's White Mountain streams would be the most practical and effective means to reduce warming in the reaches we studied. Buffer strips along streams would not use an undue amount of water, and the type of vegetation to plant differs depending on microclimate and management needs. Large conifers were most effective at providing shade, but smaller trees, such as alders and willows, grew faster and could provide shade more rapidly. Plant species’ resilience to a warming, drying climate should be considered to maximize success. We found that shading was also widely reported as a promising means for lowering the temperatures of streams in other regions. Stream temperature modeling can provide a starting point for identifying site-specific management alternatives to cool streams. Additional research and development of model routines to incorporate evaporation, stream fragmentation, and other effects of climate change in addition to water temperature warming are needed to fully develop the usefulness of management tools in modeling site-specific management strategies for climate change.

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