DEVELOPMENT OF A PREDICTIVE TOOL TO ASSESS STREAM TEMPERATURE IMPACTS OF RIPARIAN VEGETATION MANAGEMENT IN A DRIFTLESS AREA TROUT STREAM

By

Ben Sellers

A thesis submitted in partial fulfillment of the requirements for the degree of

Master of Science

(Agroecology/Environment and Resources)

at the

UNIVERSITY OF WISCONSIN-MADISON

2024

Table of Contents

Figures:

Commented [BS4]: Add tables and figures from appendix to this table of contents.

Tables:

Abstract

Rising water temperatures driven by climate change threaten culturally and economically important salmonid fisheries throughout the Upper- Midwest. Unsuitable thermal regimes threaten the effectiveness of habitat restoration projects in the region, thus strategies for mitigating peak summer stream temperatures are of interest to state and non-profit fisheries managers. Using a process-based stream temperature model, this study explores the thermal impact of riparian tree planting and tree removal in a 178.75 km^2 watershed in the unglaciated Driftless Area of southwestern Wisconsin. By creating hypothetical riparian vegetation scenarios and systematically adding and removing woody vegetation from the banks we explore the influence of shade and channel geometry on July stream temperatures with an emphasis on salmonid thermal suitability. We used this model to analyze an 18.5 km study reach to identify management areas that have the most potential to buffer downstream water temperatures throughout the summer with added shade. We developed the downstream thermal change (DTC) metric to measure the magnitude and downstream distance of temperature change following stream alterations. Modeled tree planting scenarios decreased the maximum July maximum weekly average temperature (MWAT) and July maximum weekly maximum temperature (MWMT) within an 18.5 km study area by 0.52 ℃ and 0.53 ℃ respectively. This study offers a workflow that is accessible to stream restoration managers using free and open-source modeling tools and common data collection practices to determine the thermal impact of restoration and prioritize future management efforts in cold water stream ecosystems.

Acknowledgements

Many thanks to my advisors, Eric and Caroline, for fueling my curiosity and for introducing me to the worlds of stream ecology and watershed management. Your encouragement and patience have been an incredible gift that I have been so thankful for in the five years since we've met. Eric, our weekly check-ins have been the most beneficial learning environment I could have imagined. Your ability to make even the most intimidating modeling questions accessible and exciting has brought the best out of me like no other academic setting has. Caroline, your encouragement to think critically about landscapes and the human processes that shape them has added so much depth and nuance to my understanding of natural resource management. I would not be the person I am without you two and am proud to carry your teachings with me.

Thank you, Liz Kozik (https://liz.kozik.net/), for the beautiful artwork that has helped me in telling the story of my research in writing and presentations.

Turbulence, self-doubt, and coding frustration was quickly forgotten with the help of bike rides, river trips and walks in the arboretum with my partner, Bobbi. Thank you for your compassion and support throughout this process.

Thank you, Sydney, for your nearly ten years of friendship, leading the way through grad school, and for always taking the extra time to poke around.

Lastly, thank you to my parents, Scott and Cindy, for your encouragement and love. Your pursuit of your passions is inspiring, as is your dedication to providing the best possible lives for your children. I'm so proud of both of you for your resilience in the face of uncertainty this past year.

Chapter One: Stream Temperature Background & Management Context

Formatted: Font: Bold, Underline

Water temperature is one of the most important variables governing water quality and overall aquatic ecosystem health in streams and is threatened by increasing air temperature related to climate change and human driven impacts (Ficklin et al., 2023). Thermal regimes in rivers are controlled by a suite of biotic and abiotic variables including climate, geography, geologic setting, interactions with terrestrial ecosystems, and alteration of watershed processes by human development (Caissie, 2006). The alteration of stream temperature regimes can lead to changes in biogeochemical processes and has reduced distributions of thermally sensitive aquatic species globally (Coutant, 1999).

Water temperature is inversely proportional to dissolved oxygen concentration, which is often a limit for sensitive stream species including salmonids. As oxygen becomes less available, intolerant species are forced to seek out refugia, reducing their habitat availability and increasing competition. The metabolism of instream biota increases with warming temperature, which can lead to competition for dissolved oxygen, further stressing cold water obligate species (Caissie, 2006). Additionally, increased water temperature may increase the toxicity of substances to fish and aquatic insects and increase disease vulnerability (Washington DNR). As the climate warms, the availability of cold water refugia continues to decrease, limiting the abundance of sensitive aquatic species across the globe (Van Vliet et al., 2013). The driving mechanisms of stream temperature are deeply intertwined, which complicates the ability of riparian and watershed management to mitigate changes in stream temperature related to climate change (Ouellet et al., 2020). Thermal regimes in rivers are controlled by multiple competing environmental and physical characteristics.

The complex heat budget of a stream comprises energy from solar radiation being absorbed by the stream, interaction between atmospheric and water surface (evaporation, cooling/warming from air temperature), exchange of heat between the stream and its bed surface, longwave radiation flux, and downstream heat transport (Webb & Zhang, 1997; Leach et al., 2021**)**. Another critical determinant of stream temperature is stream discharge, controlling the volume of water, its resulting thermal capacity, and the amount of energy required to warm it. Stream gradient and channel width determine the residence time of water within a reach, the extent of the air-water interface, and the height and density of riparian vegetation required to shade the channel (Moore et al., 2005). Riparian vegetation, channel orientation, and the surrounding landscape control the amount of solar energy interaction with the stream surface throughout the year (Sparrow et al., 2018; Blann et al., 2002). The degree to which each of these characteristics influence the stream's temperature response is inherently linked to chemical and biological processes in streams and is an important control of the distribution of aquatic organism assemblages including salmonids (Bray et al., 2017). The thermal regime of cold-water ecosystems is an important concern for natural resource managers due to their rich biodiversity as well as cultural and economic importance.

This is certainly true in Wisconsin, which is home to stream-dwelling salmonid populations and abundant cold-water resources, especially in the Driftless Area in the Southwest corner of the state. Surface water in the Driftless Area is dominated by spring fed streams, renowned for their biologic richness and cold temperatures year-round. Recreation is a high priority in watersheds that host game fish, namely native Brook Trout (*Salvelinus fontinalis*) and introduced Brown Trout (*Salmo trutta)*, which have been able to thrive in recent decades despite (and, in the case of brown trout, sometimes because of) large scale ecological changes brought by European settlement. The Driftless Area is a highly regarded trout fishing destination among anglers, driving a culturally and economically important tourism economy (Ross, 2013)**.** Hundreds of miles of state managed fisheries areas can be found in Wisconsin's Driftless Area, with fish densities rivaling famous western fly-fishing destinations. However, temperature rise due to climate change poses a threat to the cold-water resources of the region and the fish that depend on them.

Rising stream temperature, driven by anthropogenic climate change, has been identified as a concern among Driftless Area fisheries managers (Olson et al., 2021). Salmonids are cold blooded ectotherms which require cold water temperatures for each of their life stages (Elliot, 1994). Modeling studies have projected a 68% decrease in suitable thermal habitat for Brook Trout in Wisconsin by the mid-21st century as a result of air temperature increases associated with climate change (Mitro et al., 2019). Climate change is expected to increase average stream temperature across Wisconsin as annual air temperatures increase across all seasons (Stewart et al., 2015). Between 1950 and 2006, annual air temperature in Wisconsin increased 1.1 °F (Kucharik et al., 2010**)**; as air temperature plays a large role in the energy budget of a stream, stream temperatures are thus predicted to rise across Wisconsin. Complicating this process is the role of groundwater in moderating the warming of rivers. Precipitation has increased in recent decades and is expected to continually increase through the end of the century, especially during fall, winter, and spring, which may increase base~~-~~flow and offset some warming~~.~~ (WICCI, 2011; Deitchman & Loheide, 2012). ~~Though~~ However, groundwater temperature is likely to rise with increasing air temperatures, which may negate the thermal benefits of increased baseflow (Murdock, 2017). Stream managers are aware of these interacting drivers and interested in developing restoration strategies, including vegetation management, which can slow rising water temperatures and reduce habitat loss as a result (Olson, 2022) but often lack the modeling tools necessary to plan for and design such strategies (Schuster, 2017).

Restoration is commonly prioritized in watersheds and reaches where thermal regimes approach or exceed tolerance limits for cold water obligate organisms (Kirk Olson, personal communication). Restoration practitioners often alter in-stream hydraulics and riparian zone vegetation to ~~change~~ influence the thermal regime of rivers and streams (Fuller et al., 2022). Riparian vegetation management influences temperature regimes by altering the amount of incoming solar radiation interacting with the stream surface (Hall & Selker, 2021). Channel narrowing and subsequent deepening through stream restoration and engineering practices is also effective in altering temperature regimes by increasing water velocity and reducing the residence time within a reach, while decreasing stream surface area subject to warming by incoming radiation (Merriam, 2019; O'Brian, Shephard, & Coghlan 2017). Driftless Area streams have experienced extensive alterations to promote the current distribution of fish habitat in the face of unprecedented degradation in the late 19th and early 20th centuries.

Widespread conversion of prairie and Oak Savanna to pasture and row crops by European settlers in the mid-19th century led to dramatic changes in the hydrology and geomorphology of Driftless Area streams. Bare soil on steep slopes due to overgrazing and cropping reduced infiltration and accelerated erosion through increased surface runoff during large rainfall events; this led to widespread sedimentation in Driftless Area valleys. The resulting shift in hydrologic regime caused stream channels to widen, migration rates to increase, and stream banks to heighten due to overbank deposition (Knox, 2019). A tenfold increase in floodplain sedimentation rate from pre settlement prompted increases in bank height that caused widespread disconnection between area streams and their floodplains (Knox, 2006). Adoption of soil conservation practices in the early 20th century was widely successful in slowing sedimentation rates and restoring infiltration (Juckem et al., 2008). Contour strips, crop rotations, and conversion of steep hillslopes from row crops to pasture and forest were some of the practices in part responsible for this shift. Restoration of baseflow from soil conservation practices and an increase in annual average precipitation dramatically improved stream conditions for cold water obligate salmonids, which had been locally extirpated in many watersheds. In tandem with fish stocking and habitat management by state fisheries managers, these practices paved the way for Driftless Area streams to become the renowned resource they are today.

Stream restoration in the Driftless Area has primarily been intent on enhancing trout habitat in streams for the development of a recreational fishery, and to stabilize property against the erosive impacts of increasingly large floods (Trout Unlimited, 2023). Many stream channels which were wide and slow moving in the early to mid-20th century, have been narrowed, and fish habitat for adult brown trout has been ~~supplemented~~ improved by providing overhead cover and the winnowing of fine sediment that reveals spawning gravel through the installation of in-stream habitat structures and the mechanical slope reduction of steep banks (Vetrano, 2019). In-stream habitat restoration efforts in the Driftless Area, combined with farm scale conservation practices which improved groundwater recharge and restored baseflow, has led to recognition as a nationally renowned recreational fishery, driving recreational tourism which bolsters local economies by generating an estimated $1+ billion dollars annually (Anderson, 2016). Many stream restoration projects in the region involve replacement of riparian trees, mainly boxelder (*Acer Negundo*), with cool season grass mixtures in an effort to reduce local erosion and improve

Commented [SL9]: Could cite Eric's work here

Commented [EB10R9]: I wouldn't worry about this.

angler access (Wisconsin DNR, 2023). The effect of this restoration style on stream temperature has been a topic of intense debate.

Managers in the Driftless Area hold competing perspectives about the utility of riparian vegetation for attenuating peak summer water temperatures. The importance of shade provided by riparian trees is often countered by concern for increases in channel width related to accelerated bank scour within forested reaches (Blann et al., 2002; Gottschalk Druschke et al. 2023; Lundberg et al. 2022**)**. Additionally, fallen trees can limit angler access, while overhead branches can be a nuisance to fly angling (Lyons et al., 2000). The influence of recreational angling on stream character is non-trivial, as restoration projects are mainly funded by fishing license sales, trout stamp add-ons, and Trout Unlimited fundraising (Trout Unlimited, 2023; Wisconsin DNR, 2023)**.** Further complicating the management of these streams is the fact that a vast majority of land within the region is privately owned and managed for agricultural production. However, through landowner agreements between state Natural Resource Management agencies and land trusts, many watersheds have a considerable amount of public fishing easements, which grant access to anglers and DNR management activities, including vegetation and habitat management (Wisconsin DNR, 2002). The role of streamside vegetation in controlling thermal and geomorphic processes has been the subject of multiple research efforts in the region (Trimble 1997; Cross et al., 2013; Gaffield et al., 2005)

While riparian vegetation reduces solar radiation interacting with the stream surface, it also impacts channel width to depth ratio, which is an important driver of stream temperature. One study, using a statistical approach, observed grassy streams have, on average, a 67-72% lower width to depth ratio compared to forested reaches in the Coon Creek watershed (Trimble,1997). This trend may be a product of shade limiting understory vegetation colonization, resulting in local erosion of bare stream banks (Lyons et al., 2000). Using a process-based modeling approach, Blann et al. (2000) found that in a Driftless Area stream, areas with channel widths less than 2.5m with wooded buffers had no advantage over grass buffers in decreasing water temperatures while in wider reaches, wooded riparian zones significantly reduced summer average and maximum temperatures. Additional modeling studies by Gaffield et al. (2005), explored the effect of simultaneous channel widening and shading from riparian forests, reporting that channel widening by 50% would offset the local cooling effect of riparian shade. Modeled shade increases in a Central Wisconsin brook trout stream were observed to increase trout habitat by 4.9 km compared to current conditions (Cross et al., 2013). Vegetation management and channel alterations are important tools for restoration practitioners to manage stream thermal regimes, though interactions between vegetation and channel shape can lead to complex stream temperature responses. Stream temperature modeling provides a platform to consider the many complicating factors controlling summer water temperatures in Driftless Area streams, and the ability of restoration to make meaningful improvements to stream temperature regimes on a scale that is relevant to fisheries management.

Using a process-based modeling approach, we address the growing interest among stream restoration and fisheries managers regarding the thermal effects of riparian vegetation in Driftless Area streams by providing spatially explicit results of restoration thermal outcomes. Within our model we examine the effectiveness of tree planting and channel widening as well as tree removal and subsequent channel narrowing to improve thermal habitat for salmonids during peak summer stream temperatures. By simulating vegetation and geomorphic scenarios we aim to provide support for stream restoration efforts to prioritize the improvement of salmonid habitat, angler access, erosion mitigation and property protection, while improving the resilience of stream temperatures to future climate warming.

Chapter Two: Development of a Predictive Tool to Assess Stream Temperature Impacts of Riparian Vegetation Management in a Driftless Area Trout Stream

(For submission to River Research and Applications)

Abstract

Rising water temperatures driven by climate change threaten culturally and economically important salmonid fisheries throughout the Upper- Midwest. Unsuitable thermal regimes threaten the effectiveness of habitat restoration projects in the region, thus strategies for mitigating peak summer stream temperatures are of interest to state and non-profit fisheries managers. Using a process-based stream temperature model, this study explores the thermal impact of riparian tree planting and tree removal in a 178.75 km^2 watershed in the unglaciated Driftless Area of southwestern Wisconsin. By creating hypothetical riparian vegetation scenarios and systematically adding and removing woody vegetation from the banks we explore the influence of shade and channel geometry on July stream temperatures with an emphasis on salmonid thermal suitability. We used this model to analyze an 18.5 km study reach to identify management areas that have the most potential to buffer downstream water temperatures throughout the summer with added shade. We developed the downstream thermal change (DTC) metric to measure the magnitude and downstream distance of temperature change following stream alterations. Modeled tree planting scenarios decreased the maximum July maximum weekly average temperature (MWAT) and July maximum weekly maximum temperature (MWMT) within an 18.5 km study area by 0.52 ℃ and 0.53 ℃ respectively. This study offers a workflow that is accessible to stream restoration managers using free and open-source modeling

tools and common data collection practices to determine the thermal impact of restoration and prioritize future management efforts in cold water stream ecosystems.

Introduction

Water temperature is one of the most important variables governing water quality and overall aquatic ecosystem health in streams and is threatened by increasing air temperature related to climate change and human driven impacts (Ficklin et al., 2023). Thermal regimes in rivers are controlled by a suite of biotic and abiotic variables including climate, geologic setting, interactions with terrestrial ecosystems, and alteration of watershed processes by human development (Caissie, 2006). The alteration of stream temperature regimes can lead to changes in biogeochemical processes and has reduced distributions of thermally sensitive aquatic species globally (Coutant, 1999).

As oxygen becomes less available with increasing water temperature, intolerant species are forced to seek out refugia, reducing their habitat availability and increasing competition (Caissie, 2006; Washington DNR). As the climate warms, the availability of cold water refugia continues to decrease, limiting the abundance of sensitive aquatic species across the globe (Van Vliet et al., 2013). The driving mechanisms of stream temperature are deeply intertwined, which complicates the ability of riparian and watershed management to mitigate changes in stream temperature related to climate change (Ouellet et al., 2020).

The complex heat budget of a stream comprises energy from solar radiation being absorbed by the stream, interaction between the atmosphere and water surface (latent and sensible heat flux), exchange of heat between the stream and its bed surface, longwave radiation flux, and downstream heat transport (Webb & Zhang, 1997; Leach et al., 2021). Stream channel geometry is a critical determinant of stream temperature. Channel depth controls the amount of energy required to heat a stream reach, while channel width determines the amount of surface area available for energy exchange as well as the height and density of riparian vegetation required to shade the channel (Moore et al., 2005). Water velocity determines the residence time of water within a reach and the rate of downstream advection. The degree to which each of these characteristics influence the stream's temperature response is inherently linked to chemical and biological processes in streams and is an important control of the distribution of aquatic organism assemblages including salmonids, which is an important concern for natural resource managers due to their rich biodiversity as well as cultural and economic importance (Bray et al., 2017).

Climate change is expected to increase average stream temperature across Wisconsin as annual air temperatures increase across all seasons (Stewart et al., 2015: Kucharik et al., 2015). Fisheries modeling studies have projected a 68% decrease in suitable thermal habitat for Brook Trout (*Salvelinus fontinalis)* in Wisconsin by the mid-21st century because of air temperature increases associated with climate change (Mitro et al., 2019). However, precipitation has increased in recent decades and is expected to continually increase through the end of the century, especially during fall, winter, and spring, which may increase base~~-~~flow and offset some warming (WICCI, 2011; Deitchman & Loheide, 2012). ~~Though~~ However, groundwater temperature is likely to rise with increasing air temperatures (Murdock, 2017). Stream managers are aware of these interacting drivers and interested in developing restoration strategies, including vegetation management, which can slow rising water temperatures and reduce habitat loss as a result (Olson, 2022) but often lack the modeling tools necessary to plan for and design such strategies (Schuster, 2017).

While riparian vegetation reduces solar radiation interacting with the stream surface, it also impacts channel width to depth ratio, which is an important driver of stream temperature. One study in the unglaciated Driftless Area of Wisconsin found that grassy streams have a 67-72% lower width to depth ratio compared to forested reaches, on average (Trimble,1997). This trend may be a product of shade limiting understory vegetation colonization, resulting in local erosion of bare stream banks (Lyons et al., 2000). Using a process-based modeling approach, Blann et al. (2000) found that in a Driftless Area stream, areas with channel widths less than 2.5m with wooded buffers had no advantage over grass buffers in decreasing water temperatures while in wider reaches, wooded riparian zones significantly reduced summer average and maximum temperatures. Additional modeling studies by Gaffield et al. (2005), explored the effect of simultaneous channel widening and shading from riparian forests, reporting that channel widening by 50% would offset the local cooling effect of riparian shade. Modeled shade increases in a Central Wisconsin brook trout stream were observed to increase trout habitat by 4.9 km compared to current conditions (Cross et al., 2013). Restoration managers in the Driftless Area have conflicting viewpoints on the role that riparian vegetation, specifically grasses versus trees, plays in determining temperature, with both sides citing scientific evidence that favors their preference (Gottschalk Druschke et al., 2023). Additionally, the role of riparian shade may be overlooked by stream restoration projects which prioritize benefits of grassy riparian zones because of their tendency to promote stream narrowing, and improved angler access. Further clarification is needed to determine how riparian vegetation can best mitigate high summer stream temperatures in Driftless Area trout streams while balancing pressure from anglers, agriculture, and climate change.

Stream temperature modeling provides a platform to consider the many complicating factors controlling water temperatures in rivers, and the ability of restoration to make meaningful improvements to temperature regimes on a scale that is relevant to fisheries management. Statistical stream temperature modeling approaches require less data than process-based models, though are less effective at representing changes in specific drivers of stream temperature like riparian vegetation and climate (Dugdale et al., 2017) Process-based temperature models enable researchers and managers to explore the dynamics of interactive drivers of stream temperature by simulating management scenarios and quantifying potential outcomes prior to costly restoration efforts, though require more data (Hall & Selker, 2021). While multiple studies offer modeling approaches to quantify changes in stream temperature because of riparian tree planting, there are few that use this approach to prioritize restoration based on management relevant scenarios. Additionally, there is a lack of watershed scale process-based temperature modeling studies focused on the unique hydrologic setting and management context of Wisconsin's Driftless Area.

Considering the concerns of Driftless Area restoration managers and gaps in the existing literature, we asked: (**1) How does summer stream temperature change following riparian forest thinning and subsequent stream narrowing along a single stream reach? (2) How does summer stream temperature change across a watershed's heterogeneous stream network under various riparian management scenarios? (3) Within the stream network, where would introduction of riparian forests be more/less effective at reducing increases in summer stream temperatures?** In response to these questions, we designed and implemented a process-based modeling approach, leveraging field observations from summer 2023 in a Driftless Area watershed. Through this modeling we address the growing interest and need among stream restoration and fisheries managers regarding the thermal effects of riparian vegetation in Driftless Area streams by providing spatially explicit results of restoration thermal outcomes.

Methods

Modeling Approach

To disentangle the relationship between riparian vegetation and other drivers of the temperature regime in ~~a~~ 4th and 5th order Driftless Area stream reaches, we leveraged in-situ and remotely sensed observations to calibrate and validate a process-based stream temperature model. Temperature logger, high resolution satellite imagery, LiDAR data, and streamflow measurements are the basis of our calibration and validation methods, which were collected in the summer of 2023. Process-based temperature models require a large amount of data to inform physics-based calculations of the mechanisms governing stream temperature, which we sourced from field data collection and publicly available databases. After using our observational dataset to calibrate and validate the stream temperature model, we simulated vegetation management scenarios that isolated the effects of vegetation and channel geometry on stream temperature to better understand spatial patterns useful in informing management. Additionally, using hydraulic and vegetation conditions observed in summer 2023, we developed an index of restoration priority to create recommendations of riparian plantings and predictions of their effects on stream temperature as it relates to salmonid thermal preferences.

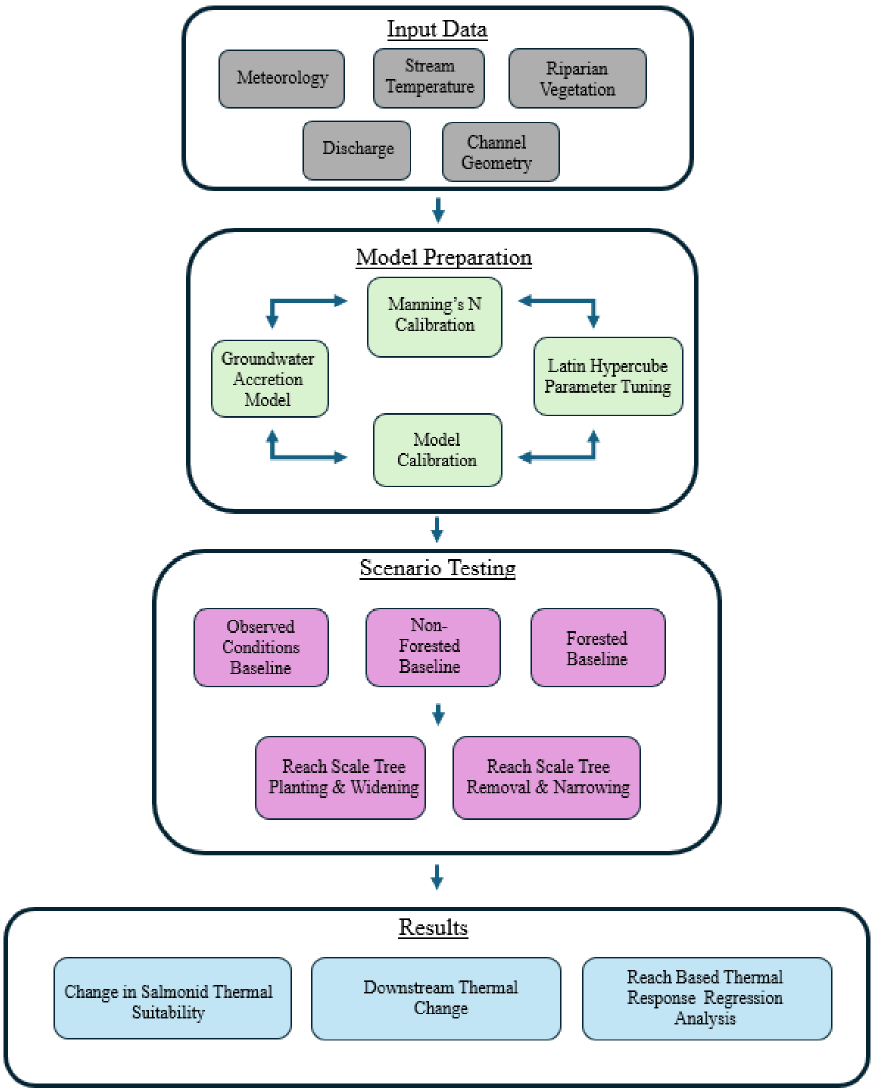

Figure 1: Schematic diagram of modeling workflow.

Study Area

Data collection was centered on 43 sites along 19.1 km of the West Fork of the Kickapoo River (WFK) widely known for its value as a recreational trout fishery. The watershed is located in Southwestern Wisconsin, within the unglaciated Driftless Area. Driftless Area streams have high base-flows relative to the rest of the state (Gebert et al., 2011). This high baseflow is partly a result of high recharge rates on surrounding hillslopes, and local hydro stratigraphy which consists of alternating layers of highly porous (sandstone, dolomite) and impermeable bedrock layers (shale) (Potter, 2019). Bedrock heterogeneity causes a large amount of groundwater discharge variability in the watershed, resulting in a non-linear pattern of stream warming moving downstream.

The WFK watershed is a HUC-10 unit measuring ~305 km^2 in area. The land cover of the watershed is predominantly agricultural with 52% of the land area in pasture and cultivated crops, followed by deciduous forest cover (38%), according to the National Land Cover Dataset (NLCD) (Dewitz, 2021). Bedrock heterogeneity causes a large amount of groundwater discharge variability in the watershed, resulting in a non-linear pattern of stream warming moving downstream (Potter, 2019).

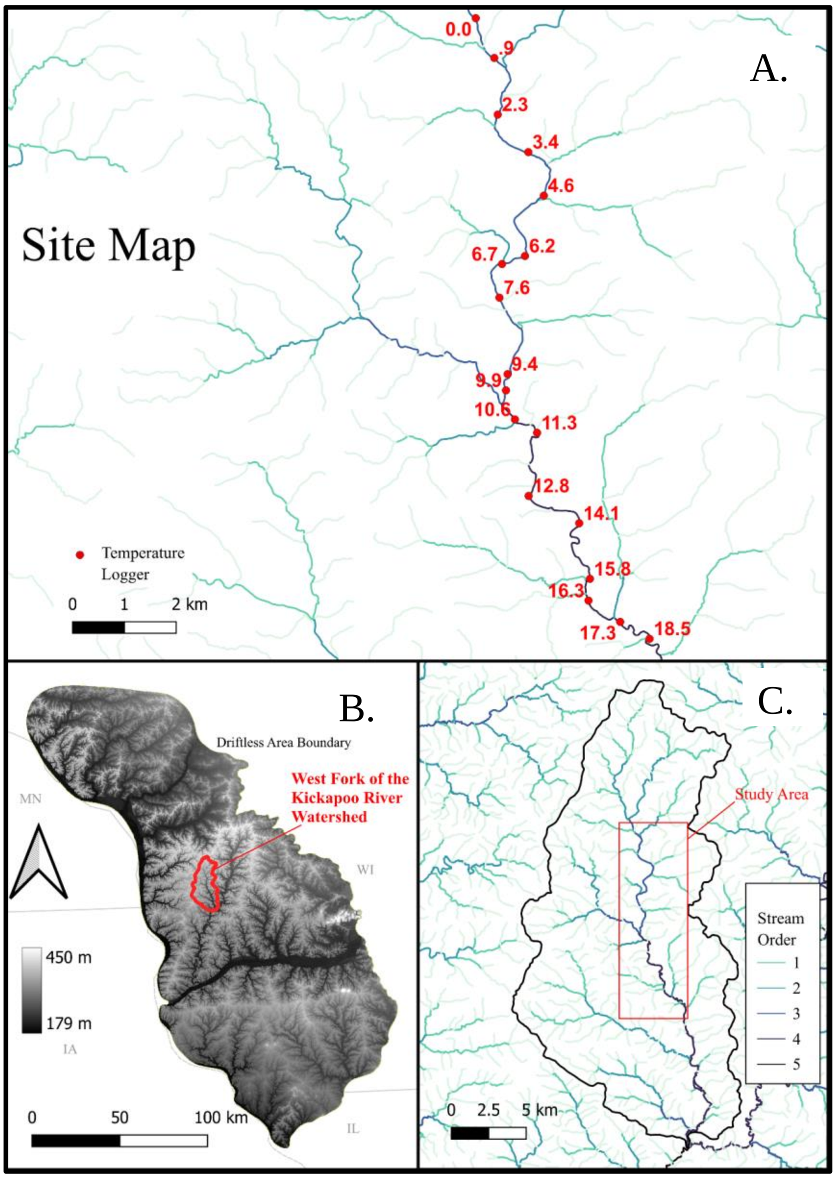

Figure 2: Project site map and geographic context.

Stream Temperature Model

To investigate the effects of stream restoration techniques including riparian tree planting, tree removal, and alterations to channel shape on the thermal regime of the WFK River, this study utilized Heat Source version 9.0.0b26, a publicly available, open-source, process-based stream temperature model (Michie, 2022). Heat Source simulates open channel hydraulics, flow routing, heat transfer, and effective shade to estimate hourly stream temperature at discrete nodes evenly spaced throughout the study reach (Boyd & Kasper, 2003). Heat Source was originally developed to monitor the role of riparian forest removal in salmon streams of the Pacific Northwest and has been used in state and federal total maximum daily load (TMDL) determination processes to manage water quality (Dadoly & Michie, 2010). Heat Source is implemented using Python 3.8 and includes a one dimensional hydraulic and heat transfer model to determine stream temperature using the primary drivers both within the stream channel and surrounding riparian zone. Model inputs include boundary temperature and flow conditions, groundwater accretion volume and temperature, stream shading, channel geometry, and meteorological conditions. Integration with ArcMap 10.8 allows the use of GIS and remote sensing data as model inputs through TTools software, a collection of Python scripts dedicated to acquiring vegetation height and channel geometry measurements. Heat Source model routines are run at equally spaced nodes along the stream centerline. Our model spans 18.5 km of the upper WFK watershed, with 100m node spacing. Using finite difference methods, hourly stream temperature is determined by calculating the total energy flux within the stream, including evaporative, convective, longwave & shortwave radiation, and conductive fluxes using the following equation:

Commented [EB31]: Citation here for the model itself (the reference should have the github website).

Commented [BS32R31]: Doesn't seem like any other heat source studies cite the github page, just the original boyd and kasper paper. Not sure of how to cite the github page but looking into i.

Commented [BS33R31]: Point to the place where it was downloaded

Commented [BS34]: Add to bibliography

Φ Total = Φ Evaporation +Φ Convection +Φ Longwave +Φ Solar +Φ Conduction

(Equation 1)

The total energy flux is used to calculate stream temperature change using:

$$\frac{\partial T}{\partial t} = -U \cdot \frac{\partial T}{\partial x} + Dl \cdot \frac{\partial^2 T}{\partial x^2} + \frac{\Phi Total}{D * cH20 * m} \quad \text{(Equation 2)}$$

Where U represents average flow velocity, T is water temperature, x is distance downstream, Dl is the dispersion coefficient, D is the average stream depth, cH20 is the specific heat capacity of water (4,182 Jkg−1*C−1), m is the density of water, and dt is the model time step.

The model includes equations for determining incoming longwave & shortwave radiation accounting for riparian vegetation, bank shade, and topographic features. The one-dimensional hydraulic model that is provided within Heat Source implements the Muskingum-Cunge method to simulate flow routing and volume storage at each model node, providing inputs to calculate heat fluxes within the stream and resulting in hourly stream temperature predictions (Oregon DEQ, 2006).

Model Inputs

Observational data

To create a continuous stream temperature calibration and validation dataset for use in modeling the effects of restoration, we deployed a network of 30 temperature loggers across our 19.3-km study reach in the WFK watershed, including mainstem and major tributaries. Stream temperature was measured using HOBO Water Temperature Pro V2 Data Loggers (Onset Computer Corporation) anchored to the stream bed from May to October 2023, recording water temperatures continuously every 15 minutes.

In areas where incomplete tributary records exist, time series imputation was used to extend temperature records using the 'simputation' package in R (Van Der Loo, 2024). Linear models were created using the 'impute_lm' function to extrapolate the relationship between available tributary data and the two nearest continuous tributary records with similar contributing areas and timing of peak temperature. Resulting tributary results were compared to observed values to evaluate the strength of the relationship.

Meteorological data were collected using a HOBO U30 weather station operating at the lower boundary of the study area. Measurements of wind velocity, relative humidity, air temperature, and solar radiation were continuously logged at a 15-minute interval throughout the field season and resampled to hourly observations for modeling purposes. All sensors were mounted 2 m above ground level.

Synoptic stream discharge measurements were collected at 44 sites across the study area, including mainstem and tributary sites. Discharge measurements were taken within 100 m of continuously logging temperature sensors. Observations were collected using a SonTek Flowtracker 2 handheld acoustic doppler velocimeter (Xylem Inc.). Collection was focused on periods of low streamflow to monitor base-flow characteristics for use in hydraulic and temperature modeling. Sites with minimal upstream and downstream obstructions (vegetation,

Commented [BS36]: Do I need a citation for this in the bibliography?

Commented [EB37R36]: Nope

boulders, bridge pylons) were prioritized for Flowtracker measurements to ensure accurate flow characteristics.

Channel width measurements were collected during discharge measurements and used to verify manual GIS digitization of high-resolution leaf off satellite orthoimagery provided by Planet Labs (Planet Team, 2017), with reference to publicly available bare earth LiDAR digital elevation models (DEM).

Streamflow & Groundwater Discharge

Without a continuous record of stream discharge in the watershed, we used a nearby USGS gauge in the adjacent Kickapoo River watershed (Gauge ID:05408000) to estimate the WFK mainstem and tributary discharge for times without direct observation using the ratio of WFK discharge to the daily average Kickapoo River discharge. The discharge records for the upstream boundary condition and tributary inflows were extended by applying this proportion to daily average flows from the nearby USGS gage.

Based on work from Schuster (2017) and Juckem (2006) that showed the strong variability in recharge across physiographic zones (ridge, hillslope, valley), we developed a method that estimates groundwater discharge based on the upstream contributing area's composition of those physiographic zones and estimated recharge rates for each zone (Figure A4, Figure A5). Annual recharge rates were found to correlate closely with the amount of each physiographic class by Juckem et al. (2006**)** through the development of a groundwater flow model in the neighboring Coon Creek watershed (Table 1).

Physiographic Type	Recharge Rate (cm/year)
Valley	20.1
Hillslopes	36.3
Ridges	16.0

Table 1: Recharge rates based on physiographic regions from Juckem (2006) developed for neighboring Coon Creek Watershed.

Groundwater temperature was measured at the valley bottom in a shallow groundwater well near the downstream boundary of the study area and used to create bounds for which to calibrate the model parameters, outlined in the model calibration section.

Vegetation

Leaf-off LiDAR from 2020 was used to create a canopy height model in ArcGIS Pro representing height above a normalized ground surface using by a classified point cloud (WROC, 2020). Tree crown polygons were created using the marker-controlled watershed segmentation function in the ForestTools R package, which segments tree crowns based on treetop points determined by a local maxima of raster values (Figure A6) (Plowright, 2023).

Commented [EB41]: I would add a brief sentence describing how this tool works.

Commented [BS42R41]: Tried to do this

Model Calibration and Validation

Hydraulic Calibration

With hydraulics – and the channel characteristics derived from it such as width: depth ratio – being an important driver of the temperature regime in a stream reach, we first calibrated the 1-D stream hydraulics component of Heat Source by varying Manning's N values across the domain to ensure accurate representations of observed channel width and depth values. We used a single parameter Latin hypercube sampling approach with Manning's N values ranging between .015 and 0.15 and calculated resulting errors in the width:depth ratio. Using these results, the Manning's N value which minimized the RMSE between modeled and observed width:depth ratios was assigned to each of the 16 nodes where width:depth observations were recorded in the field. Points between observed measurements were estimated through linear interpolation between the nearest upstream and downstream points with calibrated manning's N values.

Temperature Calibration and Validation

Following hydraulic calibration, we calibrated and validated the temperature component of our WFK Heat Source model using three separate week-long time periods in late June, early July, and early August. Morphological inputs, wind function coefficients, accretion temperature, and canopy cover variables were varied using value ranges informed by field observations and similar modeling studies (Bond et al., 2015, Dugdale et al., 2020, Spanjer et al., 2022). Parameters were iteratively sampled throughout a realistic range of values using a Latin hypercube approach with 1000 realizations for each calibration period. Results of each model run were compared to downstream observed temperature loggers to determine hourly RMSE values at each sensor location. The RMSE of each sensor across the 3 calibration periods were combined and used to create a randomized 70/30 calibration/validation split. The parameter set which resulted in the lowest overall calibration RMSE was selected as the final model values for use in the Heat Source model scenarios.

Model Scenarios

To model the result of potential restoration activities on stream temperature, two categories of incremental, reach-based restoration scenarios were modeled: idealized and management-focused. In our idealized model runs, two uniform riparian vegetation conditions were used as baselines – forested and non-forested – to examine the effect of incremental riparian tree removal (from forested baseline) and planting (from non-forested baseline) on 500-m reaches throughout the study area to analyze shade effectiveness without confounding upstream and downstream shading effects. In management-focused model runs, the same process of iteratively changing reach shade through tree planting was applied but observed vegetation conditions (observed baseline) were used across the study area as a baseline to incorporate the spatial heterogeneity of existing shade conditions and to provide manager-relevant effectiveness metrics.

For the idealized scenarios, forested baseline conditions were created by replacing current vegetation with a uniform tree canopy of 7 m. The canopy polygon extends out from both banks by 10 m and overhangs each bank by 3 m vertically. Modeled tree heights represent an early successional forest type, though tree heights were varied in our sensitivity analysis (described below). In the non-forested baseline, all vegetation was replaced with 0.7-m tall grasses, based on field observations (Figure 3). Canopy closure values were varied as a part of model calibration.

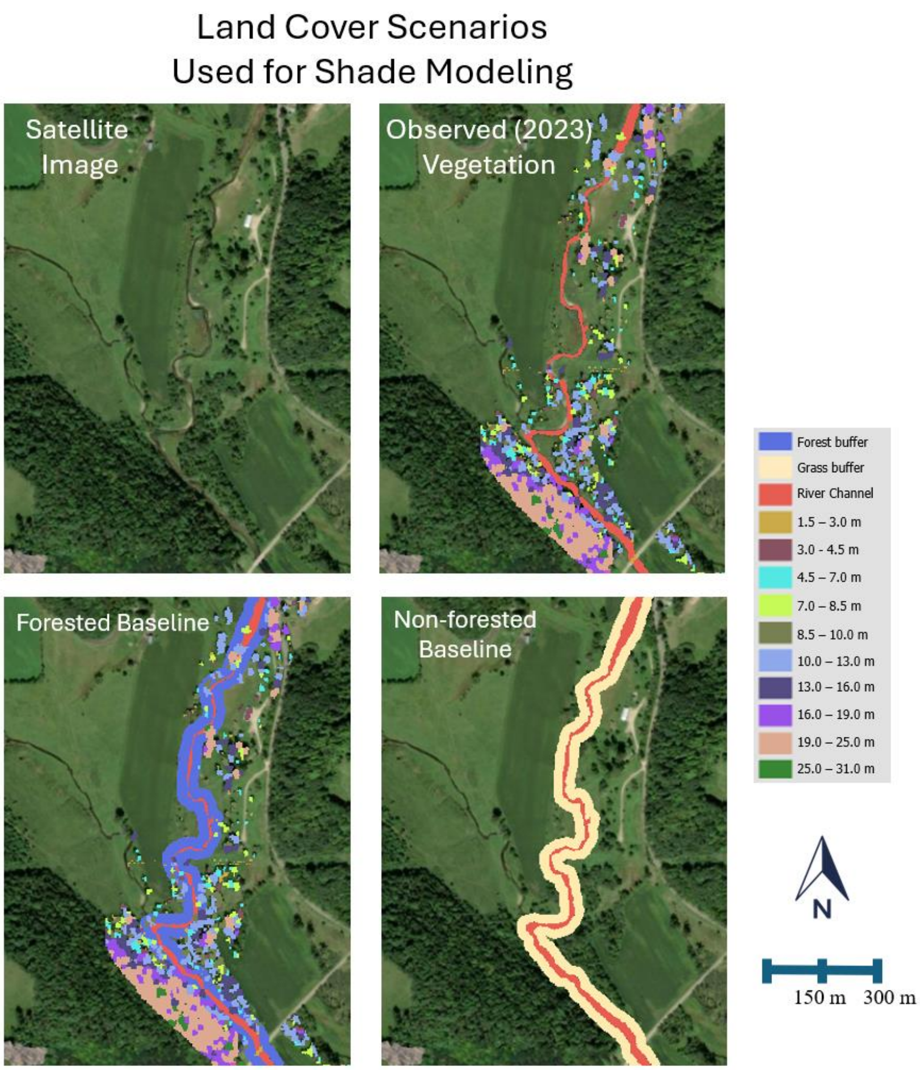

Figure 3: GIS inputs from landcover scenarios.

Within idealized baseline scenarios, channel widths were altered to represent geomorphic differences observed in previous studies that identified significant channel widening in forested reaches within the Driftless Area (Trimble, 1997, Lyons et al., 2000). In the non-forested baseline, all nodes that were forested within the observed vegetation conditions were narrowed by 20%. Nodes were determined to be forested if they had an average effective shade > 20% with effective shade defined as the percentage of incoming potential solar radiation that does not reach the stream surface. To create the forested baseline, each non-forested node was widened by 20% (Figure 4). This decision was also based on an analysis of channel widths in forested and grassy streams throughout the study area, which found a trend of increased width in reaches with forested riparian zones (Figure A1).

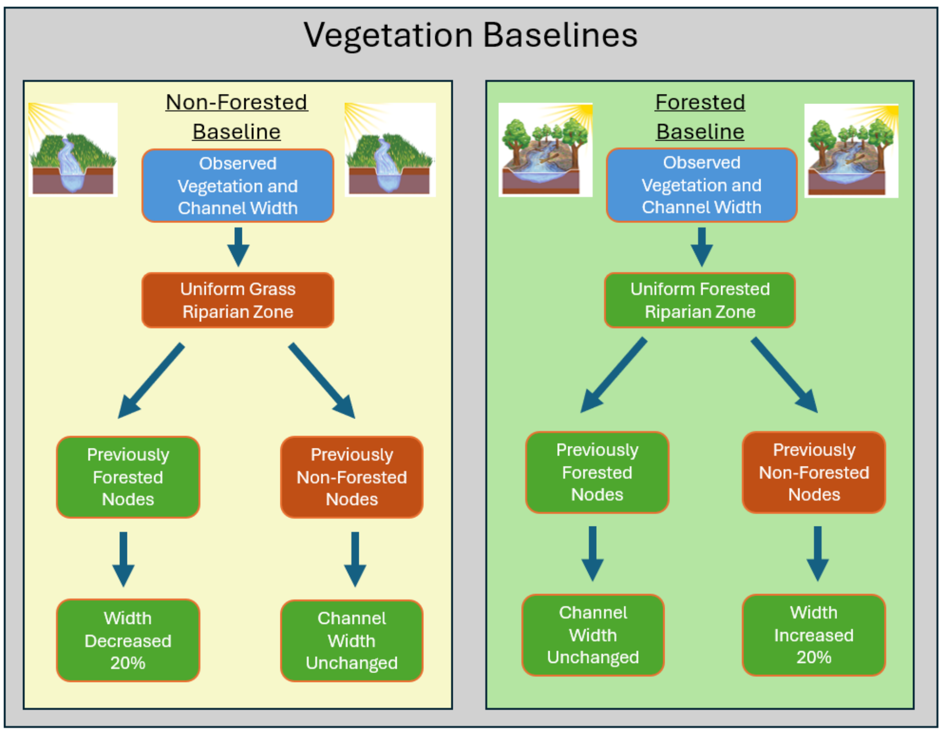

Figure 4: Vegetation baselines developed for model scenarios.(Illustration credit: Liz Anna Kozik)

Within each idealized scenario we converted a 500-m reach, resembling a possible management unit, from trees to grass or grass to trees, depending on the base scenario. Model scenarios using all-grass (non-forested) and all-forest (forested) baselines were simulated by incrementally converting 500-m reaches from the baseline to the alternate condition and narrowing or widening depending on the transition (i.e., narrowing in forest to non-forest, widening in non-forest to forest) to determine spatial patterns where each vegetation type would be most effective at lowering summer stream temperatures across the watershed. In the observed baseline tree planting scenario, 7-m tall woody vegetation is added incrementally across 500-m reaches, in addition to existing woody and non-woody vegetation. Within the 500-m restoration reach, any nodes that are not forested were widened by 20% and currently forested nodes were left at their observed width (Figure 5; Table 2).

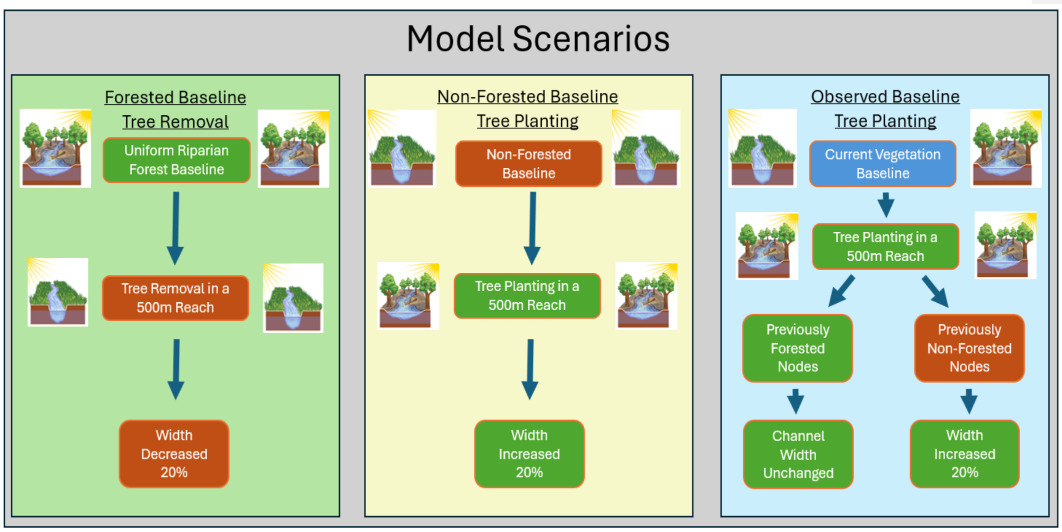

Figure 5: Heat Source model scenarios. (Illustration credit: Liz Anna Kozik)

Scenario	Baseline Vegetation	Baseline Channel Width	Vegetation Changes	Geomorphic Changes
Observed Baseline – Tree Planting	Observed (2023) vegetation	Observed channel width (orthophoto)	Replace existing vegetation with uniform (7m tall) riparian forest in a 500m reach	Widening channel 20% at previously non-forested nodes within 500m reach
Forested Baseline - Tree Removal	Uniformly forested riparian zone	Previously non-forested nodes widened 20%	Remove existing forest vegetation and replace with grasses in a 500m reach	Narrowing channel 20% within 500m reach
Non-forested Baseline – Tree Planting	Uniform grassed riparian zone	Previously forested nodes narrowed 20%	Replace existing grass vegetation with uniform (7m tall) riparian forest in a 500m reach	Widening channel 20% within 500m reach

Table 2: Overview of Heat Source model scenarios

In addition, the downstream boundary of the temperature model was extended by 5 km using average of channel width and gradient values from the final 1 km. Landcover conditions for the 5 km extension are the same as the observed in the final downstream node of the study area. This allows for changes at the downstream end of the study to continue downstream past the boundary of our observational data set to avoid bias towards upstream restoration scenarios, which affect more of the study area than downstream sites.

Commented [EB60]: Something weird is going on with the spacing. Maybe try moving Figure 6 a little bit up or down a paragraph?

We created the Downstream Thermal Change (DTC) metric to compare the effectiveness of restoration in each 500-m reach along the study area to change downstream temperatures, positively or negatively, by integrating between the longitudinal temperature profiles of mean July water temperature at each node across the stream network before and after an alteration (Figure 6). The resulting value represents the area between the two scenarios, which captures both the magnitude of temperature differences as well as the downstream distance that temperature change is observed. A negative DTC value corresponds to a net cooling effect on downstream temperatures, while a positive DTC value refers to downstream warming.

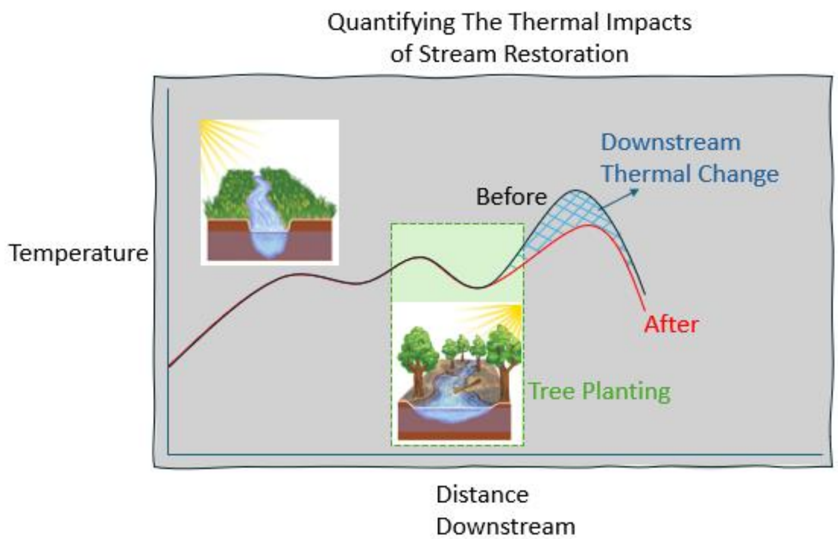

Figure 6: Example of Downstream Thermal Change (DTC) metric. (Illustration credit: Liz Anna Kozik)

The magnitude of change downstream from each incremental 500-m reach alteration was used to rank which reaches are most sensitive to thermal impacts of increased and decreased solar radiation and associated channel width change. DTC was used to investigate the effects of velocity, channel width, width:depth ratio, and change in solar radiation on stream temperature following restoration scenarios.

The same workflow was used with observed baseline conditions to develop prioritization maps and projected thermal benefits from increasing riparian shade within the study reach. With the observed conditions (2023) as the baseline, a layer of 7-m vegetation, representing an early successional forest buffer, was added uniformly in 500-m increments. Existing vegetation that exceeded 7-m was left within the experimental reaches, to reflect realistic restoration in which tree planting takes place without removal of the current riparian canopy. DTC values associated with each 500-m alteration provide a prioritization index for future vegetation management.

We then simulated an aggregated management scenario in which the 10 reaches where shade is most effective are altered with additional vegetation to estimate the potential of a 5-km shade restoration effort to offset summer peak stream temperatures with native salmonids in mind. Temperatures were evaluated based on 7-day averages of daily mean and maximum temperatures to determine the resulting changes to salmonid thermal habitat from riparian afforestation. The weekly rolling average of daily average and maximum temperatures are commonly used temperature metrics for classifying chronic and acute temperature exposure to salmonids, without bias toward daily peak temperatures (Carter, 2005)**.** Wehrly et al. (2007) identified thermal critical maxima for brook and brown trout of 23.3 ℃ for 7-day average mean temperature (7-d MEANT) and 25.4 ℃ for 7-day average maximum temperatures (7-d MAXT) based on analysis of brook and brown trout distributions in 285 stream sites in Michigan and Wisconsin.

We anticipated that tree height and channel width changes (resulting from vegetation changes) would be strong drivers of stream temperature therefore we simulated them over a range to investigate their impact on modeled temperatures. The amount of widening and narrowing taking place in the all-forested and all-grass baselines was compared at 20%, 30%, and 40% (Figure B4, Figure B6). Additionally, the height of the forest buffer in the all-forested baseline scenario was compared using 7m, 9m, and 11m tall forest buffers (Figure B6).

Commented [EB66]: Following on Caroline's comment, this does seem a bit arbitrary. Anything to add to firm up this decision?

Commented [BS67R66]: Not going to remove trees in order to plant trees

Results

Hydraulic Calibration

Latin hypercube stratified sampling was used to select a Manning's N value for each model node with an adjacent width and depth measurement. Manning's N values ranged from 0.015 to 0.15 allowing for changes in channel depth as needed to maintain observed width:depth ratios (Figure 7). Absolute errors of width:depth ratio between modeled and observed values ranged from 0.25 to 48. Larger errors occurred in areas where the appropriate Manning's N to fit model depths to observed depths were out of the range of tolerance decided upon by field observations and reference guides for hydraulic modeling (Chow, 1959).

Velocities that resulted from Manning's N calibration had a range between 0.035 m/s and 0.55 m/s. Simulated velocities compared favorably (RMSE = 0.126 m/s) to observations collected during discharge measurements and are reported in Appendix 2.

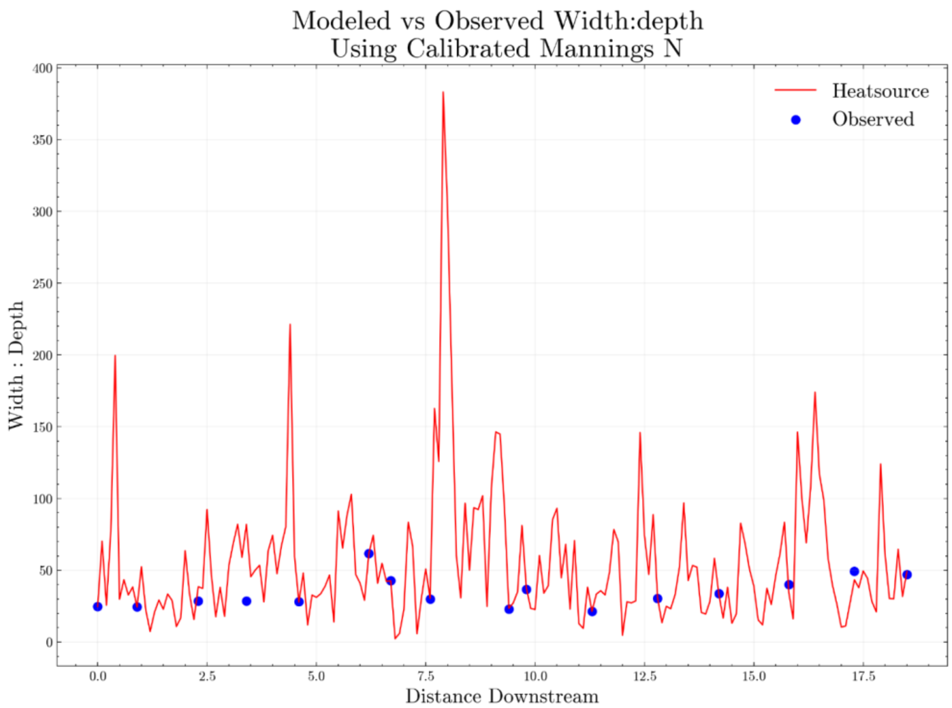

Figure 7: Modeled and observed channel width:depth ratio.

Temperature Calibration

The calibrated parameter set (Table 3) led to an overall average RMSE for the calibration dataset of 0.62℃, and 0.63 ℃ for the validation dataset. These results have similar accuracies reported by past modeling studies that used the Heat Source temperature model (Bond et al., 2015; Deitchman & Loheide, 2012; Spanjer et al., 2022) (Figure B7). Overall averages of hourly RMSE throughout the calibration period was prioritized as an evaluation metric to ensure the model represents the overall patterns and drivers in stream temperature throughout the calibration period, rather than prioritizing peak daily temperature, or daily average.

Parameter	Calibration Range	Final Value
Sediment Thermal Conductivity (W/m/C)	1.4- 1.6	1.51
Sediment Thermal Diffusivity (cm^2/second)	6.2 x 10^3 – 6.8 x 10^3	6.4 x 10^3
Hyporheic Thickness (m)	.24 - .4	.26
Hyporheic Exchange Percent (Decimal Fraction)	.01 - .4	.01
Sediment Porosity (Decimal Fraction)	.6– 1.0	.92
Wind Function Coefficient a	1.0 x 10^9 - 4.0 x 10^9	2.97 x 10^9
Wind Function Coefficient b	1.0 x 10^9 - 4.0 x 10^9	1.08 x 10^9
Accretion Temperature (°C)	9.0 – 13.0	10.21
Tree Canopy closure (Decimal Fraction)	.5 - .8	.79
Grass Canopy Closure (Decimal Fraction)	.1 -.2	.13
Tree Overhang (m)	0 - 2	.94
Grass Overhang (m)	0 - 1	.48

Table 3: Heat Source calibration parameters and final values.

Calibrated Temperature Model Output

WFK temperature was simulated for the month of July using observed meteorological conditions, upstream boundary conditions, and tributary inputs. Flow conditions were kept constant throughout the month-long temperature simulation, corresponding to flows used in the July calibration period. July maximum values of 7-day rolling average of mean and max temperature (7-d MEANT & 7-d MAXT) are used to summarize areas where thermal regime exceeds tolerances observed in previous trout habitat studies in the region both in chronic and acute temperature exposure (Figure 8, Figure 9) (Wehrly et al., 2007).

Maximum modeled July 7-d MEANT does not exceed 23.3 ℃ within our modeled period. However, previous studies of brook trout thermal tolerance in Wisconsin have used 22.3 ℃ as an upper thermal tolerance, reported by Eaton et al. (1995), which is exceeded in the lower half of the study area. Modeled values for maximum July 7-d MAXT exceeded the trout thermal limit of 25.4 ℃ in 1.3 km of the study area. The distribution of model nodes exceeding 7-d

MAXT temperature thresholds is concentrated in the upper and middle reaches of the study area, between model km 0.0 and 0.6, as well as between model km 9.6 and 10.1.

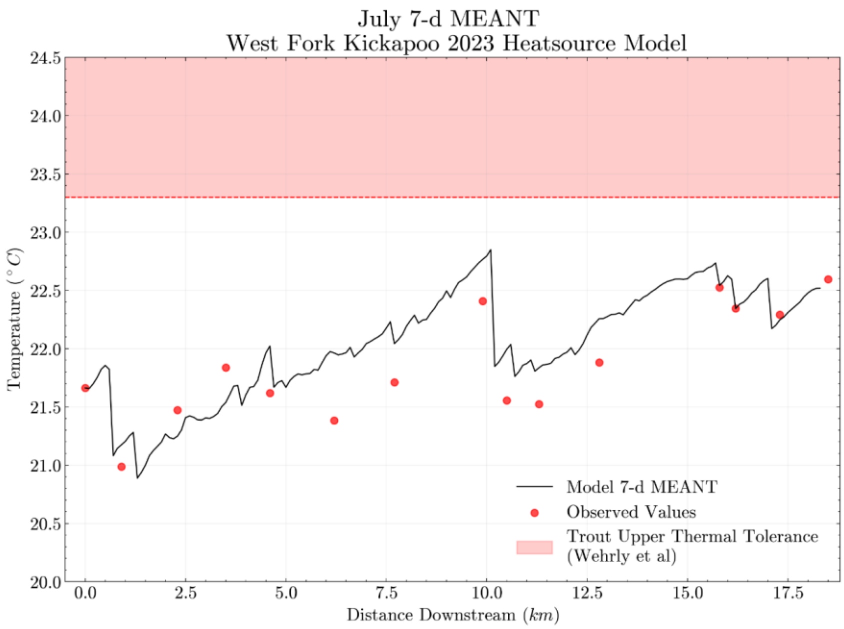

Figure 8: July maximum of 7-day rolling average of daily mean temperatures.

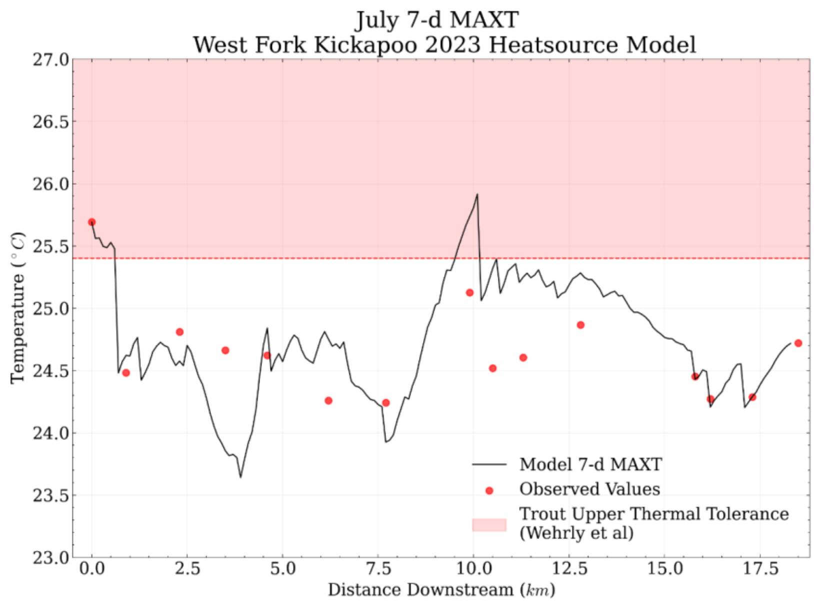

Figure 9: July maximum of 7-day rolling average of daily maximum temperatures.

Model Scenarios

Non-Forested Baseline - Tree Planting

Relationships between shade and channel metrics and downstream thermal change following incremental tree planting in the non-forested baseline varied in strength (Figure 17). There is a strong trend in this scenario between temperature reductions and the change in solar radiation reaching the stream (R^2= .75). Width may be an indicator of the sensitivity of a reach to increases in shade. The relationship between mean reach width and DTC is statistically significant ($p<.05$), and many of the widest reaches are less sensitive to tree planting. Headwater reaches, which are narrower on average **(**Figure A1**)**, can be shaded by shorter vegetation heights than wider streams and therefore tree plantings may be more effective at reducing summer water temperatures in narrower reaches.

Using the non-forested baseline, the relationship between shade increase from tree planting and channel orientation~~,~~ was found to be statistically insignificant ($p > .05$). While width is also a large determinant of the effectiveness of tree planting to increase shade, many of the least sensitive reaches to tree planting were oriented East to West (Figure 12).

The most sensitive reach to additions of shade and widening occurred at model km 3.0, with a DTC of -1.78 ℃-km. It experienced an increase in mean effective shade of 39.5%, from 10.9% to 50.4% across the reach through tree planting. The mean width of the reach after widening associated with forest establishment is 12.9 m. The mean velocity is 0.14 m/s and the width:depth ratio is 41.3. The reach that was least sensitive to cooling from additional shade occurred at model km 17.0 with a DTC of -0.13 ℃-km. This reach has a mean width of 16.3 m, a mean velocity of 0.15 m/s and a width:depth ratio of 30.24. This site experienced a relatively small change in effective shade from tree planting, from 3.75% to 22.35%, which may be a product of its East - West Orientation. As shown in our sensitivity analysis, the thermal benefits of tree planting and widening diminish as widening increases, while narrowing has the opposite effect (Appendix B4, Appendix B5).

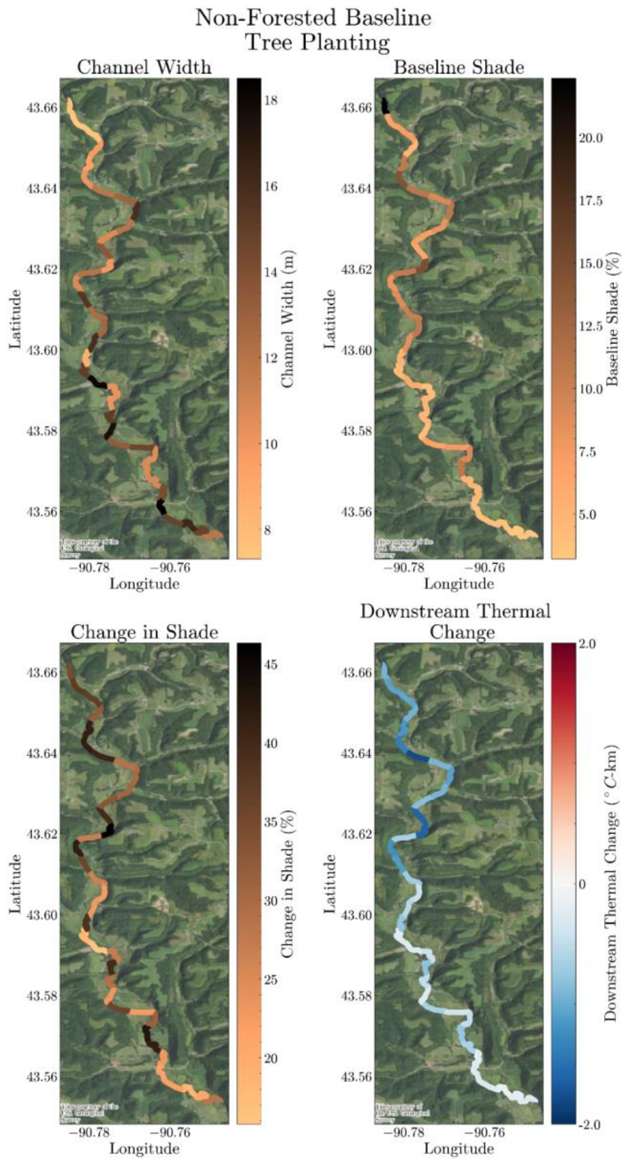

Figure 10: Non-Forested Baseline - Tree Planting Scenario: spatial output.

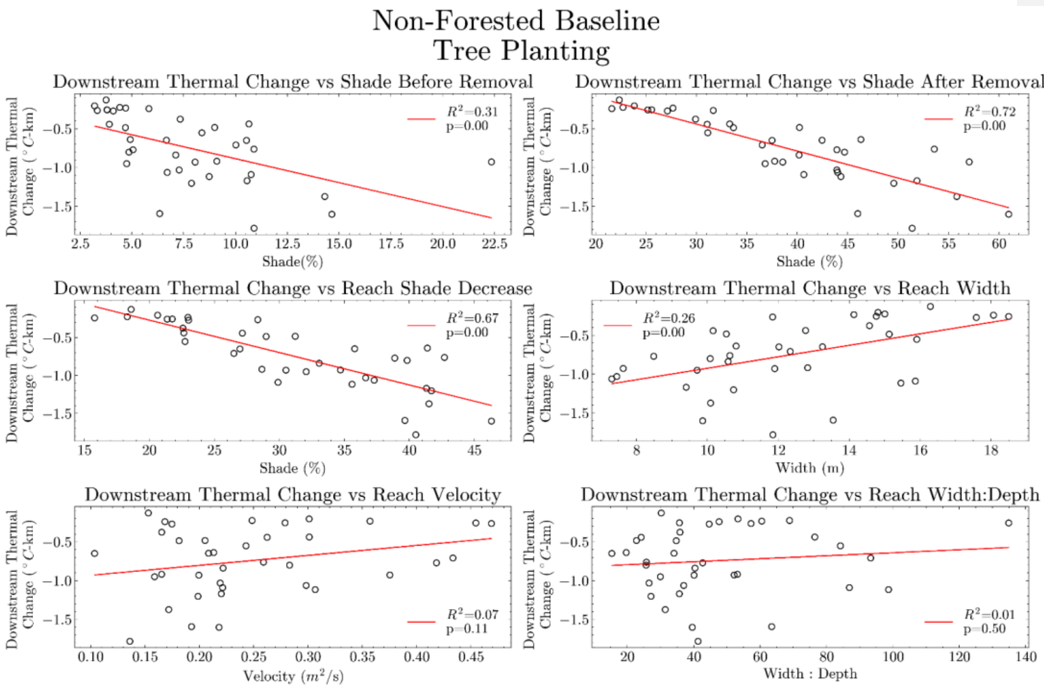

Figure 11: Non-Forested Baseline – Tree Planting Scenario: Comparison of Downstream Thermal Change to stream physical properties.

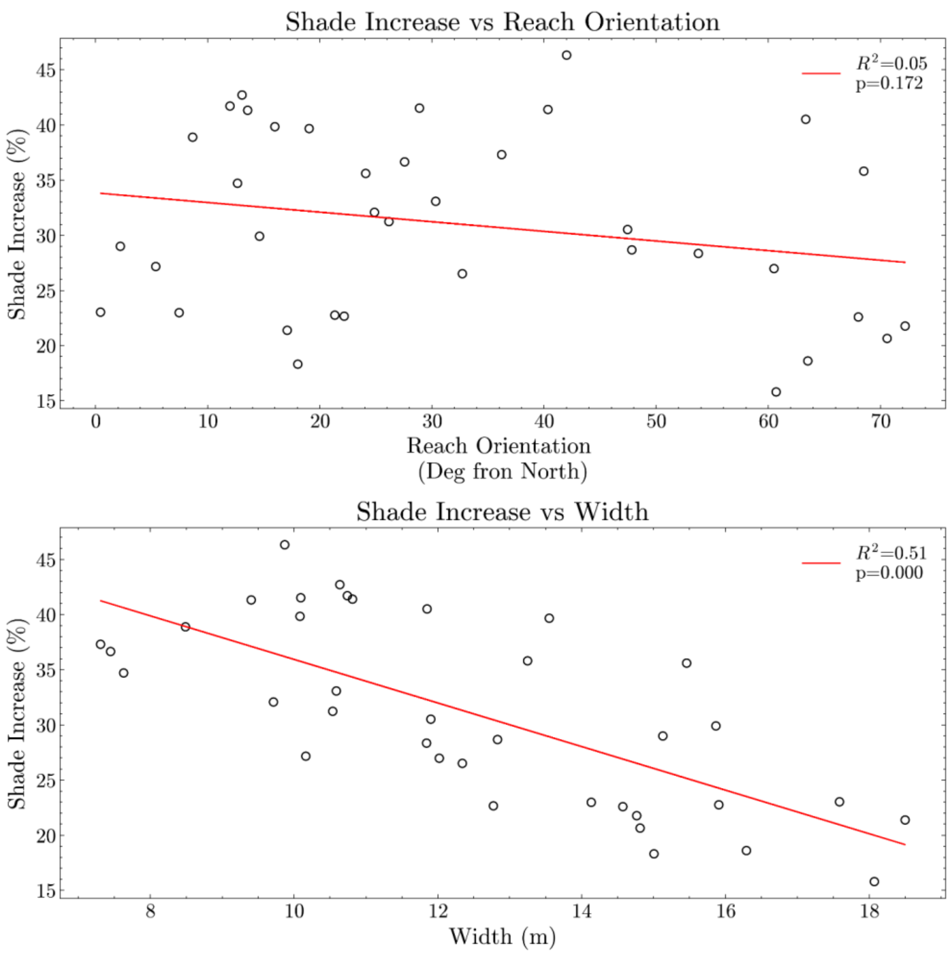

Figure 12: Impacts of channel width and reach orientation on shade increase, Non-Forested Baseline - Tree Planting Scenario

Forested Baseline - Tree Removal

In multiple reaches tree removal and narrowing resulted in low, but positive, DTC values indicating slight warming. The sites with the least capacity for altering downstream thermal regimes were seen in mid watershed and downstream reaches (Figure 14). The reaches for which tree removal and narrowing had the least effect on DTC have a low baseline shade value, indicating that forested riparian conditions in these reaches were not effective in adding shade which may be a result of their width and orientation. DTC values close to 0 in these reaches may indicate that tree removal and narrowing would be a more effective restoration technique than tree planting, especially if the channel were to narrow more than 20% from its previous width.

The highest increase in downstream temperature resulting from tree removals were seen in reaches with high baseline shade levels, which is likely a result of their narrow widths, North-South orientation and lower depth relative to higher order downstream reaches.

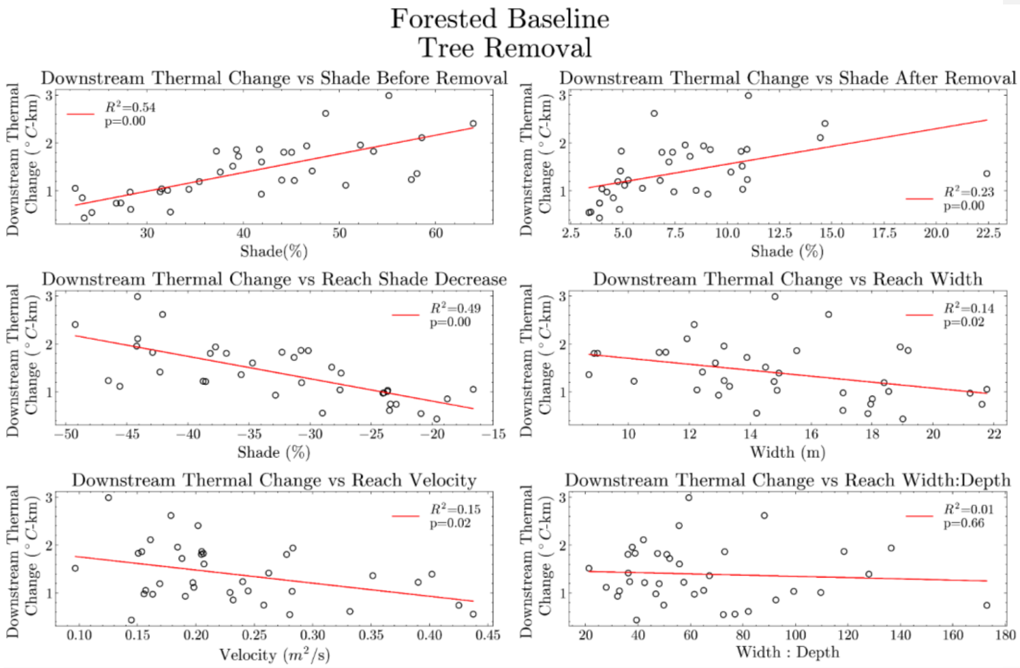

Figure 13: Forested Baseline – Tree Planting Scenario: Comparison of Downstream Thermal Change to stream physical properties.

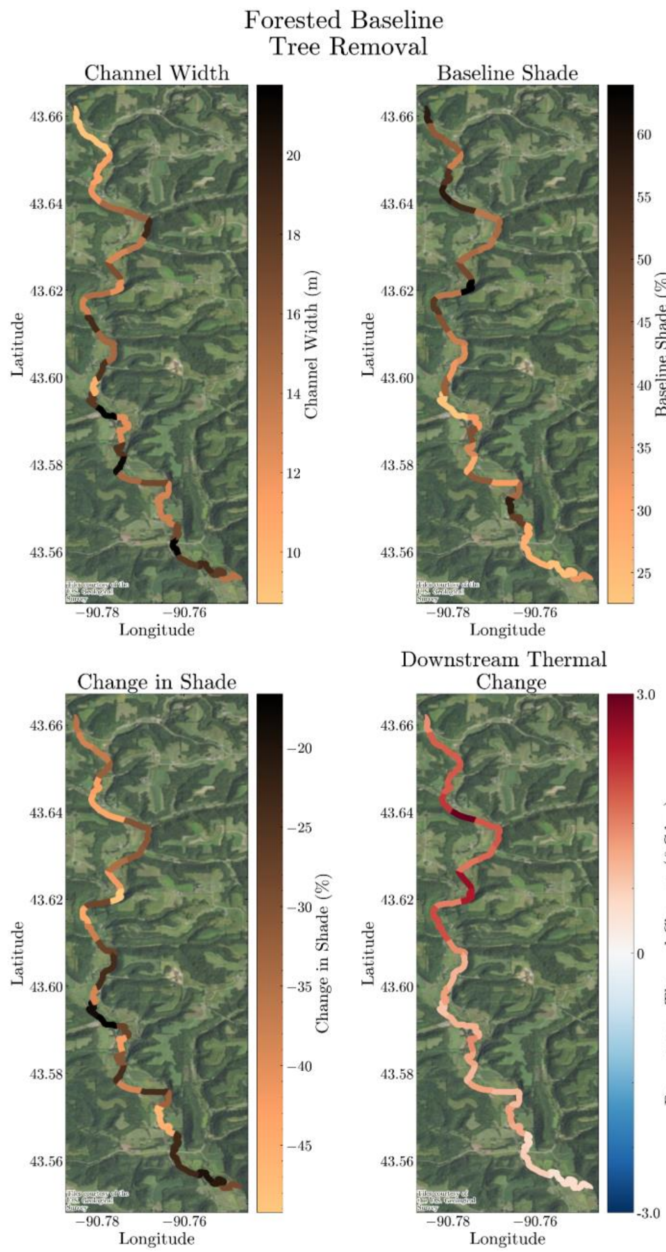

Figure 14: Forested Baseline - Tree Removal Scenario: spatial output.

Observed Baseline - Tree Planting

A separate analysis of tree planting effectiveness was done using the observed 2023 vegetation and geomorphic conditions to prioritize reaches where management could be most effective at cooling downstream temperatures (Figure 16). Thermal changes followed similar patterns as seen in non-forested and forested baseline scenarios. Increasing tall vegetation in reaches where shade is already present shows a less dramatic change in reaches which are currently shaded (Figure 15). Many upstream reaches responded to shade by cooling more than downstream reaches, which is likely a combination of channel width and lack of existing shade. This analysis provided the sensitivity of shading on an individual reach basis. Results of this analysis were used to determine the 10 reaches most sensitive to shade additions, and subsequently modeled the impact of downstream temperature regimes from tree planting on a cumulative 5 km of the study area.

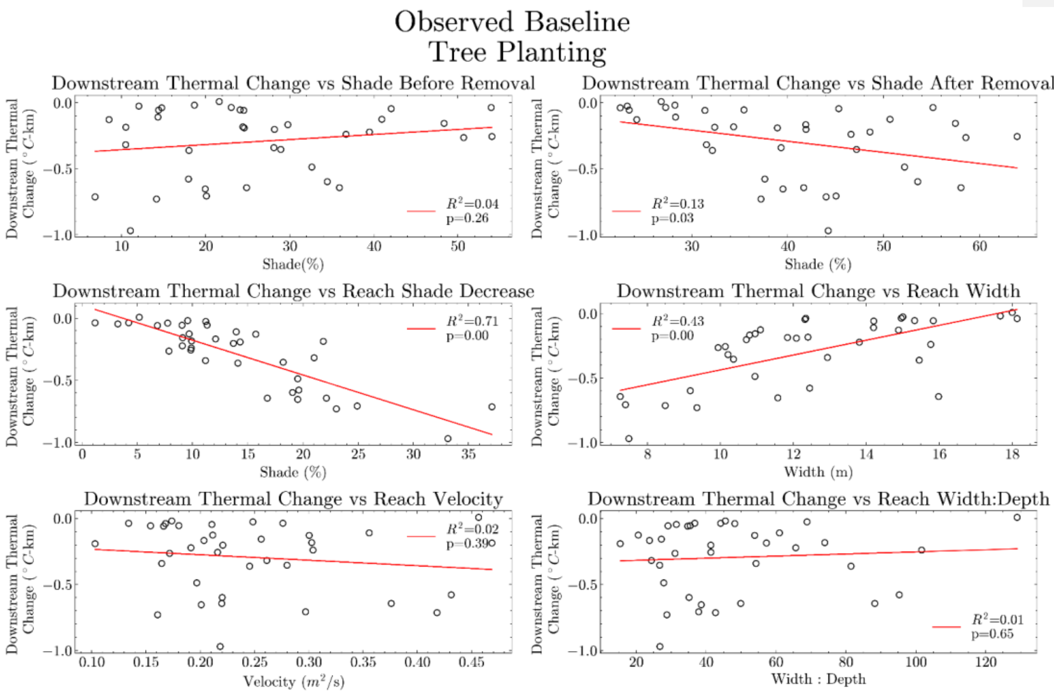

Figure 15: Observed Baseline – Tree Planting Scenario: Comparison of Downstream Thermal Change to stream physical properties.

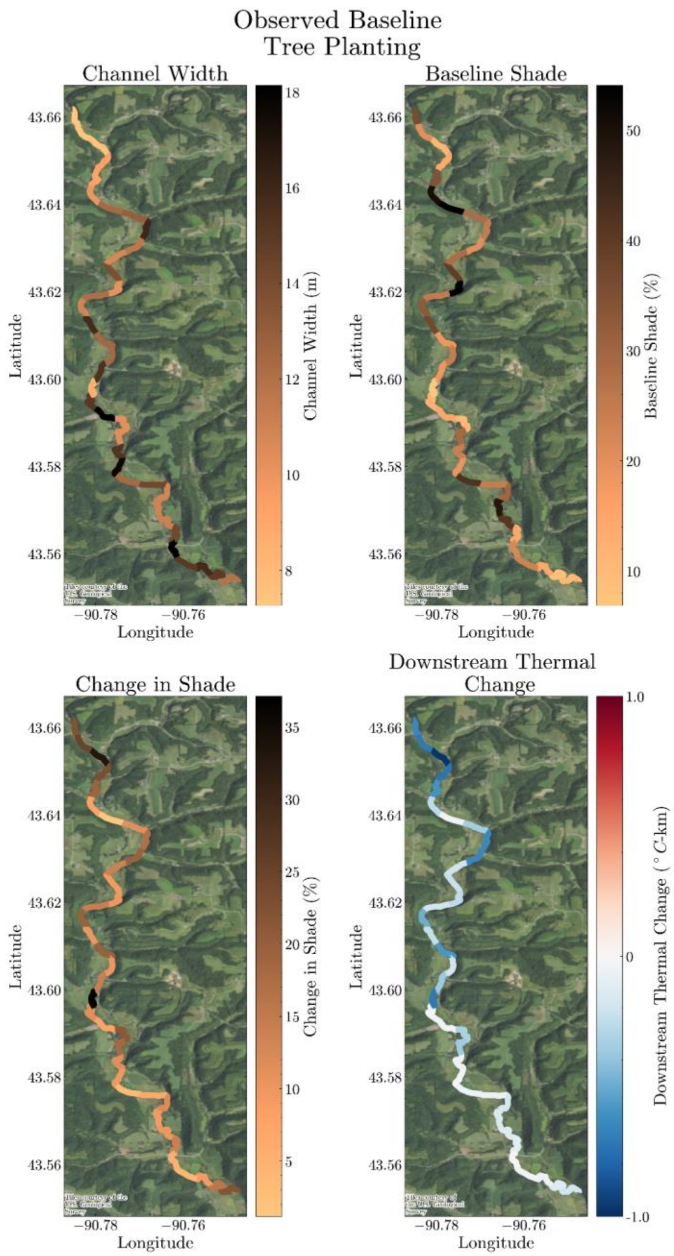

Figure 16: Observed Baseline - Tree Planting Scenario: spatial output.

Targeted Restoration Impacts

A hypothetical restoration scenario was developed from the observed baseline model, which simulates the addition of 5 km of riparian forest throughout the study area. Temperature model results indicate that the addition of a shaded riparian buffer along each 500-m reach of the stream with the 10 highest effectiveness ranks could reduce July 7-d average daily mean temperatures in this restoration scenario by a maximum of 0.52 ℃, from 22.69 ℃ to 22.16 ℃ occurring 9.9 km downstream (Figure 17). Maximum July 7-d average daily maximum temperatures were highest at model km 10.1, immediately upstream from a large tributary, and were reduced by .52 ℃ from 22.78 ℃ to 22.26 ℃ (Figure 18).

Formatted: Font color: Auto

Formatted: Font color: Auto

Formatted: Font color: Auto

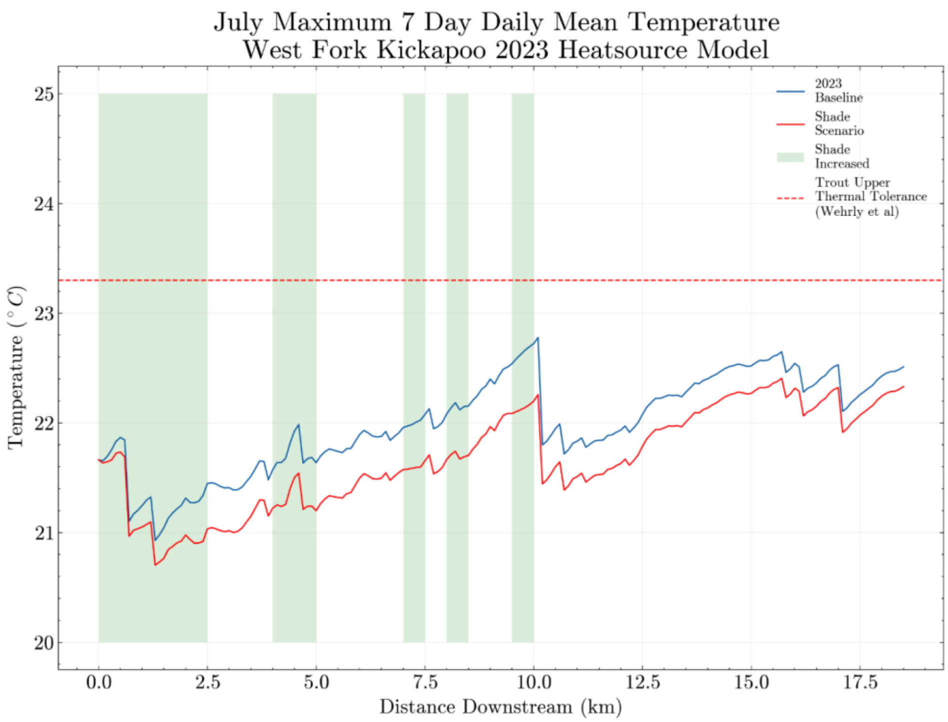

Figure 17: Thermal impacts on 7-D MAXT from tree planting on the 10 most effective management units.

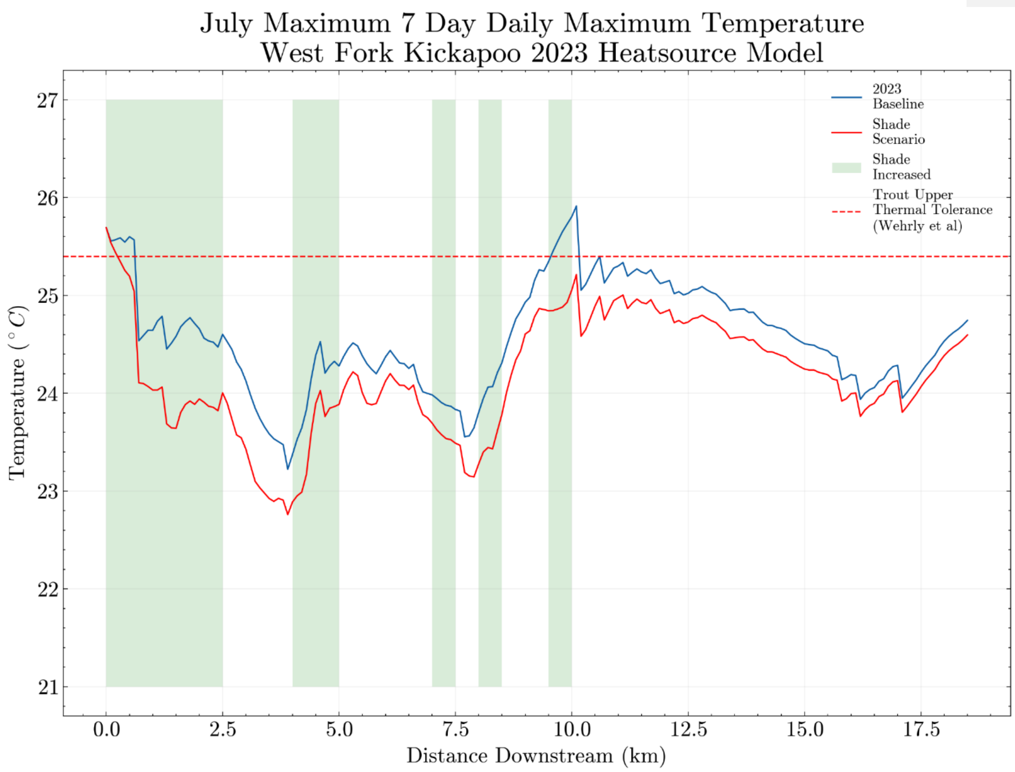

Figure 18: Thermal impacts on 7-D MEANT from tree planting on the 10 most effective management units.

Discussion

Influence of Channel Geometry, Orientation, and Topography

Stream response to riparian vegetation management is complex and dynamic, as evidenced by the heterogeneous response of modeled temperature changes following vegetation and geomorphic alterations. In narrower reaches, thermal response to tree vegetation shading is likely more sensitive because of the lower height of vegetation required to cast shade on the stream channel (Pool & Berman, 2001; Jackson et al., 2021; Blann & Vondracek, 2002). However, some narrower reaches may be more shaded by basin topography, have a higher discharge, or east-west orientation which could make the temperature response from added shade less apparent (Moore et al., 2005). In wider reaches, a warming response from tree planting and widening is possible. The magnitude of this change is determined by the height of vegetation additions and simultaneous channel widening (Figure B4, Figure B5, Figure B6).

Previous research studies have identified the width and depth of a stream channel as a strong influence on summer stream temperature at base-flow discharge (Gaffield et al., 2005; Trimmel et al., 2016; Johnson & Wilby, 2015). Gaffield et al (2005) found that channel widening associated with riparian tree cover can outweigh the reduction of shortwave radiation by shading**.** In their modeling study, thermal benefits of shade from riparian trees were offset in narrow headwater reaches when paired with a 2x increase in width. Jackson et al. (2021) reported that a 3-m wide stream with a 5-m tall, vegetated buffer would receive the same amount of solar radiation as an 8-m tall, vegetated buffer in a 5-m wide stream, and that tall trees have little benefit in narrow channels relative to shorter trees and shrubs.. In our tree planting scenarios, 20% widening associated with tree planting was decided by reference to regional geomorphic studies (Trimble, 1997). However, a 40% increase in channel width between grassed and forested reaches has been reported in Pennsylvania streams (Allmendinger, et al. 2005). In line with the findings of Li et al. (2012) we observed that as tree height was increased in a uniformly forested riparian zone, mean July water temperature steadily decreased.

In our non-forested baseline scenarios, tree planting in excessively wide reaches was less effective at increasing shade, and therefore offsetting temperatures **(**Figure 10, Figure 11**).** We also saw no significant relationship between width:depth ratio and DTC, which may be a result of averaging reach conditions over 500 m, where there could be large variation in the width:depth ratio. Combining restoration practices that both increase shade and decrease width (through additions of rigid instream structures) in wide, shallow reaches could improve the ability of riparian vegetation to reduce stream temperatures in tandem compared to employing either strategy independently.

Shade response to channel orientation followed similar patterns presented by Sparrow et al. (2018) which found vegetation more effective at shading small Driftless Area streams in N-S oriented channels compared to E-W oriented channels (Figure 12). While incoming solar radiation is higher mid-day in N-S oriented channels comparatively, the total daily flux is lower because more radiation is attenuated in the morning and evening, at lower solar angles (Li et al, 2012). In our modeled scenarios, reach orientation was not a statistically significant driver of DTC ($p>.17$), however a negative trend indicates that given a larger sample, the relationship may prove more predictive. Given this relationship, orientation should be a significant consideration for prioritization of future restoration projects in Driftless Area streams.

Additionally, topography may have an influence on the effectiveness of tree planting in offsetting stream temperatures. In multiple reaches within our study site steep valley walls abut

Commented [EB89]: Didn't see this listed either. I would go through and check to see that all references are accounted for in the References section. Are you using Zotero?

Commented [BS90]: Does this work in the discussion? Could put sensitivity analysis in the results but don't feel like its that necessary.

Commented [EB91R90]: As we chatted about today, I think it's worth moving the beginning part of this to the methods and then a little bit in results. The remainder can stay here in discussion.

the stream channel, providing a large reduction in solar radiation interacting with the stream surface. The relationship between topographic shading and DTC was significant in the non-forested baseline - tree planting scenario. Many of the reaches which have high topographic shade~~,~~ show a significant thermal response~~s~~ to tree planting. These reaches may see more benefit from riparian tree plantings compared to non-topographically shaded reaches, because topographic shade is dominated by N-S oriented ridges. The reaches which interact with these features most are also oriented N-S~~,~~ and are some of the most sensitive to tree planting.

Influence of Stream Depth

The depth of water present in a reach determines the effect of thermal inertia in moderating minimum and maximum temperatures (Trimmel et al., 2016; Johnson & Wilby, 2015). Upstream reaches with lower volume have a higher range of daily temperatures as a result compared to higher volume reaches. While the temperature pattern in lower volume upstream reaches is more influenced by groundwater discharge, vegetation has been shown to play a larger role in lowering peak temperatures in low volume headwaters than in higher order reaches (Pool & Berman, 2001). This process may in part be responsible for the significant temperature reductions in maximum temperatures observed in upstream reaches when shade was added to a previously non-forested vegetation baseline (figure 11). Our model highlights this trend, as the downstream reduction of maximum and mean temperatures were highest when shade was added to lower volume, upstream reaches and slowly decreased moving downstream, although the pattern was not linear.

Formatted: Highlight

Formatted: Highlight

Formatted: Highlight

Influence of Groundwater

Groundwater discharge and cold tributary inputs further complicate the impacts of the downstream effects of riparian shading**.** Additions of cold water from numerous tributaries throughout our study area significantly lowered stream temperatures in adjacent downstream reaches. Modeled scenarios show that areas upstream of moderate volume cold tributary inputs may be less effective in reducing downstream temperature averages. In the case of the reach between 9.0 km and 10.5 km downstream, additions of shade are less effective. This reach has the highest 7-d MEANT within the study area, and we would expect to see a large DTC here. However, this reach is directly upstream of the largest cold-water tributary in the watershed, which may be overpowering the contribution of shade from this reach. Further investigation between a restoration project's distance to the nearest tributary and its DTC could help identify potential areas for maintaining angler access with limited impact to stream temperatures. However, this may be complicated by the influence of other reach characteristics like width, channel orientation, and volume.

Effect of Shade on Salmonid Habitat

Observed and modeled mean and maximum summer water temperatures indicate significant thermal stress to salmonids within the study area. Thermal thresholds for brook and brown trout presented by Wehrly et al. (2007) acknowledge that stress associated with exposure to temperatures near and exceeding lethal limits have considerable effects on reproduction, feeding, growth, distribution and mortality. During the warmest months, brook and brown trout utilize thermal refugia including cold water tributaries, areas of high groundwater influx, and reaches with abundant overhead cover (Cross et al., 2013). Reaches with peak temperatures above the thermal threshold for trout could act as barriers to fish that would otherwise access thermal refugia (Petty et al., 2012). Increasing shade lowered weekly maximum and mean temperatures in July and reduced the exposure length of temperatures above the maximum mean temperature threshold by as much as 4 days at multiple sites (Appendix B3). Targeting restoration to mitigate excessive warming in the mainstem of the WFK could provide additional refugia to resident brook and brown trout as well as improve connectivity between cold water refugia during periods of high summer water temperatures.

Limitations/Assumptions

Because of the importance of aerial remote sensing and imagery data for development of model inputs, and the limitations associated with the frequency and timing with which this data is collected, our model is limited by relying on inputs that may no longer reflect reality. A large flood in the late summer of 2018 dramatically altered the WFK basin's geomorphology. Analysis of aerial LiDAR data which was collected in the spring of 2020 and satellite imagery collected in April 2023 may not reflect the exact channel conditions present in the summer of 2023. The most up to date, leaf-off, satellite imagery was collected during a high flow event in April 2023, thus channel measurements may differ from base-flow conditions, which could alter reach scale thermal regimes that have not been captured in the current modeling effort.

Additionally, the lack of a continuously monitoring flow gauge~~s~~ limits the ability of this model to capture fine resolution changes in discharge which might affect stream temperature dynamics. Because of the spatial heterogeneity of groundwater discharge to streams, we did not have the ability to quantify small scale thermal refugia which are known to exist in Driftless Area streams and influence salmonid habitat and thermal regime (Deitchman 2012; Dieterman et al., 2024). While this is a significant limitation, accuracies of our model are comparable to similar studies which benefit from distributed temperature sensing (DTS) observations that classify groundwater discharge and small-scale temperature heterogeneity more accurately (Wood, 2017). Additionally, the use of leaf-off LiDAR limited our model's ability to represent true vegetation height as well as if LiDAR was collected during the growing season. Leaf-off LiDAR has been observed to underestimate canopy height by 1-3m in deciduous forests (Parent, 2014). This uncertainty in vegetation canopy height has been addressed by grouping tree polygon heights based on 1.5m bins for shorter trees, 3m bins for medium trees, and 6m bins for the tallest trees (Figure A3).

Air temperature during the summer of 2023 was below the 30-year average, while mean daily streamflow was 30% less than 30 year summer averages. Streamflow observed during the study was strongly influenced by drought conditions. As a result, stream temperatures were likely higher than a typical summer. Observed streamflow conditions during the study period may simulate drought conditions that could become more frequent due to anthropogenic climate change.

Additionally, our model area does not incorporate vegetation alteration of reaches upstream of our boundary condition or tributary reaches, which may have a large impact on stream temperature. We selected the current study area because of the density of state managed easements~~,~~ and interest of stream managers in future restoration. However, further investigation into the effects of vegetation in first and second order headwater streams in the Driftless Area would benefit future management.

Management Implications

To maximize the impact of riparian tree planting on stream temperature, tree height, channel width, topographic shade, channel aspect, and local hydrology should all be considered. To achieve temperature goals, the width of the channel should be a ~~large~~major consideration when planting trees. Narrow reaches are likely the most effective reaches for riparian afforestation. Many of the widest East to West flowing reaches had little benefit from tree planting and subsequent widening. Prioritizing these reaches for tree removal and narrowing may balance managers goals of improving angler access while simultaneously maintaining or improving thermal conditions for salmonids.

An understanding of the effect of topographic shade in providing cooling effects could also play an important role in management. When considering brush and tree removal to improve angler access, prioritizing reaches with ample topographic shade could reduce warming impacts. Additionally, E-W flowing reaches have a disadvantage when it comes to shading compared to north-south flowing reaches. Since vegetation on the north sides of stream banks has little impact on stream shading, managers could maximize shade and angler access by planting trees on the south banks of east-west flowing streams and managing northern banks to improve access.

Model outputs show a clear warming trend in daily maximum temperatures within the middle of our study area and rising mean temperatures moving downstream. One potential cause of this comes from a lack of thermal inertia associated with lower discharge in upstream reaches. Where cold tributary flows enter, volume increases, and the stream is cooled due to an increase in discharge and influx of cold water. Additions of shade in deeper, higher volume areas may not be as significant as in shallow, lower volume reaches. With an increase in depth due to higher discharge, thermal inertia becomes stronger, and the magnitude of diurnal temperature fluctuations is diminished (Moore et al., 2005). Reducing temperatures in the upstream reaches using targeted tree plantings could reduce maximum temperatures seen downstream which impact salmonid feeding, reproduction, and growth (Elliot, 1994). Indirect effects of high stream temperatures, like macroinvertebrate community structure and parasite growth may have a notable impact on salmonid populations in warming streams (Dieterman & Mitro, 2019). The management and monitoring of these areas is important as the effects of climate change threaten habitat for thermally sensitive aquatic communities.

Because of the influence of temperature in determining the health and distribution of native and non-native salmonids, and the implications of climate warming on these species, the manipulation of riparian vegetation may become an important climate adaptation to preserve salmonid habitat in Driftless Area streams in the future. However, the unique hydrology of the Driftless Area requires a holistic approach when considering the role of riparian vegetation in driving stream temperature. This is especially so in comparison to streams of the Pacific Northwest, where much of the literature regarding stream temperature and stream temperature management originates. High base-flow discharge currently provides suitable thermal regimes for brook and brown trout in much of the Driftless Area, even without shade from riparian trees. Future trends in baseflow are not certain but increases in precipitation across winter and spring are predicted to increase, when the majority of infiltration and recharge occurs. Groundwater temperatures roughly follow annual average air temperatures and are likely to rise with warming air temperature (Murdock, 2017). Deitchman and Loheide (2012) saw a ~1 °C increase in stream temperature in a Driftless Area stream with a 1 °C increase in groundwater temperature using a similar Heat Source modeling approach. Considering the sensitivity of streams to baseflow temperatures and discharge dynamics, riparian shading is a critical protective measure against temperature rise in certain reaches of Driftless Area streams.

While vegetation management is an important tool for mitigating stream temperature rise, upland land use practices should not be overlooked. Groundwater aquifers are one of the most important drivers of cold summer stream temperatures, providing habitat for cold water organisms. Maintaining healthy aquifers among a changing climate will be paramount for maintaining the ecological communities found in Driftless Area streams. Infiltration is threatened by drought and changing agricultural commodity markets, facilitating increases in traditional row cropping. Agricultural practices that increase infiltration, including rotational grazing, prairie strips, and enrollment in the NRCS's Conservation Reserve Program will help preserve important groundwater resources and should be a top priority for future climate resilience of Wisconsin's cold-water fisheries.

Commented [SL103]: This content is probably better suited for the discussion as they ideas have not been developed previously in the manuscript and are not conclusions or findings of your work.

Commented [EB104R103]: I agree with Steve. We could probably cut this unless you want to put it in the discussion.

Commented [BS105R103]: I put it at the end of the discussion, does that work?

Conclusion

Concerned for the loss of sensitive salmonid habitat due to climate warming, stream restoration managers in the Driftless Area are interested in techniques for mitigating habitat loss while maintaining recreational opportunities for anglers. While grassed banks provide benefits including stream narrowing, improved angler access, and sediment storage, shade from riparian trees can reduce solar radiation interacting with the stream surface and reduce peak summer temperatures. Targeted tree planting efforts may be a necessary adaptation to rising stream temperatures, to support the region's economically and culturally significant recreational fishery. Prioritizing vegetation management to reduce summer stream temperatures, and their effect on freshwater organisms, requires considerations of a river's existing thermal regime, which is dependent on a suite of spatially explicit physical and biological characteristics. Process based temperature models provide researchers and managers with tools to understand the heterogeneity of stream temperature across a watershed, and the ability to alter these drivers through future management. These tools are useful in the Driftless Area especially, where management is limited by small easements and can provide stream managers with the ability to view large scale processes with targeted interventions.

While the management of forested riparian buffers is an important tool to improve the thermal resilience of Driftless Area streams, stream managers must balance the effects of streamside vegetation on erosion, agricultural infrastructure and angler access. As our modeling effort points out, there is no one size fits all approach to the thermal benefits of riparian shade. Given the diverse nature of stream restoration and watershed management, tailored management approaches are essential for individual sites and projects, ensuring integration with their geographic and ecological settings.

Commented [EB106]: Add this to the intro. I don't think this perspective is provided prior to this.

Commented [BS107R106]: Bottom of Page 16

Bibliography

Commented [BS108]: Formatting info for thesis: https://grad.wisc.edu/current-students/masters-guide/#what-you-need-to-do

Allmendinger, N. E., Pizzuto, J. E., Potter Jr, N., Johnson, T. E., & Hession, W. C. (2005). The influence of riparian vegetation on stream width, eastern Pennsylvania, USA. Geological Society of America Bulletin, *117*(1-2), 229-243.

Anderson, D., & Consultant, P. D. (2016). Economic Impact of Recreational Trout Angling in the Driftless Area.

Andrew Plowright (2023). ForestTools: Tools for Analyzing Remote Sensing Forest Data. R package version 1.0.0,

Bartos, M, (2020). Pysheds: simple and fast watershed delineation in python. https://github.com/mdbartos/pysheds

Belby, C. S., Spigel, L. J., & Fitzpatrick, F. A. (2019). Historic changes to floodplain systems in the Driftless area.

Blann, K., Frost Nerbonne, J., & Vondracek, B. (2002). Relationship of Riparian Buffer Type to Water Temperature in the Driftless Area Ecoregion of Minnesota. North American Journal of Fisheries Management, 22(2), 441–451. https://doi.org/10.1577/1548-8675(2002)022<0441:RORBTT>2.0.CO;2

Bond, R. M. (2015.). Sensitivity of summer stream temperatures to climate variability and riparian reforestation strategies. Elsevier Enhanced Reader. https://doi.org/10.1016/j.ejrh.2015.07.002

Boyd, M. and Kasper, B. (2003). Analytical Methods for Dynamic Open Channel Heat and Mass Transfer. Methodology for the Heat Source Model Version 7.0. http://www.deq.state.or.us/wq/TMDLs/docs/tools/heatsourcemanual.pdf,

Bray, E. N., Dozier, J., & Dunne, T. (2017). Mechanics of the energy balance in large lowland rivers, and why the bed matters. Geophysical Research Letters, 44(17), 8910–8918. https://doi.org/10.1002/2017GL075317

Caissie, D. (2006). The thermal regime of rivers: A review. Freshwater Biology, 51(8), 1389–1406. https://doi.org/10.1111/j.1365-2427.2006.01597.x

Carson, E.C. and Christensen, A.M., *in press*, Quaternary Geology of Vernon County, Wisconsin: Wisconsin Geological and Natural History Survey map, 1 plate, 1:100,000 scale.

Carter, K. (2005). The effects of temperature on steelhead trout, coho salmon, and Chinook salmon biology and function by life stage. California regional water quality control board, 1-26.

Chow, Ven Te, 1959: Open-channel Hydraulics. McGraw-Hill, NewYork. 680 p

Conservation easements | Real estate program | Wisconsin DNR. (n.d.). Retrieved April 5, 2024, from https://dnr.wisconsin.gov/topic/fl/RealEstate/easements

Coutant, C. C. (1999). Perspectives on temperature in the Pacific Northwest's fresh waters (No. ORNL/TM-1999/44). Oak Ridge National Lab.(ORNL), Oak Ridge, TN (United States).

Cross, B. K., Bozek, M. A., & Mitro, M. G. (2013). Influences of Riparian Vegetation on Trout Stream Temperatures in Central Wisconsin. North American Journal of Fisheries Management, 33(4), 682–692. https://doi.org/10.1080/02755947.2013.785989

Dadoly, J., & Michie, R. (2010). Malheur River Basin Total Maximum Daily Load (TMDL) and Water Quality Management Plan (WQMP): Oregon Department of Environmental Quality.

Deitchman, R., & Loheide II, S. P. (2012). Sensitivity of Thermal Habitat of a Trout Stream to Potential Climate Change, Wisconsin, United States1. JAWRA Journal of the American Water Resources Association, 48(6), 1091–1103. https://doi.org/10.1111/j.1752-1688.2012.00673.x

Dewitz, J. (2021). National Land Cover Database (NLCD) 2019 Products (ver. 3.0, February 2024) [Data set]. U.S. Geological Survey. https://doi.org/10.5066/P9KZCM54

Dieterman, D. J., Miller, L. M., & Hoxmeier, R. J. H. (2024). Understanding Stream-Resident Salmonid Movements in Groundwater-Fed Streams of the Driftless Area (USA). In J. Lobon-Cervia, P. Budy, & R. Gresswell (Eds.), Advances in the Ecology of Stream-Dwelling Salmonids (pp. 161–193). Springer International Publishing. https://doi.org/10.1007/978-3-031-44389-3_9

Dugdale, S. J., Hannah, D. M., & Malcolm, I. A. (2020). An evaluation of different forest cover geospatial data for riparian shading and river temperature modeling. River Research and Applications, 36(5), 709–723. https://doi.org/10.1002/rra.3598

Elliott, J. M. (1994). Quantitative ecology and the brown trout. Oxford University Press.

Ficklin, D. L., Hannah, D. M., Wanders, N., Dugdale, S. J., England, J., Klaus, J., Kelleher, C., Khamis, K., & Charlton, M. B. (2023). Rethinking river water temperature in a changing, human-dominated world. Nature Water, 1(2), Article 2. https://doi.org/10.1038/s44221-023-00027-2

Fuller, M. R., Leinenbach, P., Detenbeck, N. E., Labiosa, R., & Isaak, D. J. (2022). Riparian vegetation shade restoration and loss effects on recent and future stream temperatures. Restoration Ecology, 30(7). https://doi.org/10.1111/rec.13626

Gaffield, S. J., Potter, K. W., & Wang, L. (2005). Predicting the Summer Temperature of Small Streams in Southwestern Wisconsin1. JAWRA Journal of the American Water Resources Association, 41(1), 25–36. https://doi.org/10.1111/j.1752-1688.2005.tb03714.

Gebert, W. A., Walker, J. F., & Kennedy, J. L. (2011). Estimating 1970-99 average annual groundwater recharge in Wisconsin using streamflow data (No. 2009-1210). US Geological Survey.

Gottschalk Druschke, C., Booth, E. G., Lave, R., Widell, S., Lundberg, E., Sellers, B., & Stork, P. (2023). Grass versus trees: A proxy debate for deeper anxieties about competing stream worlds. Environment and Planning E: Nature and Space, 25148486231210408. https://doi.org/10.1177/25148486231210408

Hall, A., & Selker, J. S. (2021). High-resolution temperature modeling of stream reconstruction alternatives. River Research and Applications, 37(7), 931–942. https://doi.org/10.1002/rra.3824

Heat Source Model Analytical Framework. (n.d.). Retrieved February 15, 2024, from https://www.oregon.gov/deq/FilterDocs/MoPudAppendixCHeatSourceModel.pdf

Johnson, M. F., & Wilby, R. L. (2015). Seeing the landscape for the trees: Metrics to guide riparian shade management in river catchments. Water Resources Research, 51(5), 3754–3769. https://doi.org/10.1002/2014WR016802

Juckem, P. F., Hunt, R. J., & Anderson, M. P. (2006). Scale Effects of Hydrostratigraphy and Recharge Zonation on Base Flow. Groundwater, 44(3), 362–370. https://doi.org/10.1111/j.1745-6584.2005.00136.x

Juckem, P. F., Hunt, R. J., Anderson, M. P., & Robertson, D. M. (2008). Effects of climate and land management change on streamflow in the driftless area of Wisconsin. Journal of Hydrology, 355(1), 123–130. https://doi.org/10.1016/j.jhydrol.2008.03.010

Kalny, G., Laaha, G., Melcher, A., Trimmel, H., Weihs, P., & Rauch, H. P. (2017). The influence of riparian vegetation shading on water temperature during low flow conditions in a medium sized river. Knowledge & Management of Aquatic Ecosystems, 418, 5. https://doi.org/10.1051/kmae/2016037

Kałuża, T., Sojka, M., Wróżyński, R., Jaskuła, J., Zaborowski, S., & Hämmerling, M. (2020). Modeling of River Channel Shading as a Factor for Changes in Hydromorphological Conditions of Small Lowland Rivers. Water, 12(2), Article 2. https://doi.org/10.3390/w12020527

Knox, J. C. (2006). Floodplain sedimentation in the Upper Mississippi Valley: Natural versus human accelerated. Geomorphology, 79(3-4), 286-310.

Knox, J. C. (2019). Geology of the Driftless Area. In E. C. Carson, J. E. Rawling III, J. M. Daniels, & J. W. Attig (Eds.), *The Physical Geography and Geology of the Driftless Area: The Career and Contributions of James C. Knox* (Vol. 543, p. 0). Geological Society of America. https://doi.org/10.1130/2019.2543(01)

Kucharik, C. J., Serbin, S. P., Vavrus, S., Hopkins, E. J., & Motew, M. M. (2010). Patterns of Climate Change Across Wisconsin From 1950 to 2006. Physical Geography, 31(1), 1–28. https://doi.org/10.2747/0272-3646.31.1.1

Leach, J. A., Kelleher, C., Kurylyk, B. L., Moore, R. D., & Neilson, B. T. (2023). A primer on stream temperature processes. *WIREs Water*, *10*(4), e1643. https://doi.org/10.1002/wat2.1643

Li, G., Jackson, C. R., & Kraseski, K. A. (2012). Modeled riparian stream shading: Agreement with field measurements and sensitivity to riparian conditions. Journal of Hydrology, 428–429, 142–151. https://doi.org/10.1016/j.jhydrol.2012.01.032

Lundberg, E., Gottschalk Druschke, C., & Booth, E. G. (2022). A Q-method survey of stream restoration practitioners in the Driftless Area, USA. River Research and Applications, 38(6), 1090–1100. https://doi.org/10.1002/rra.3971

Lyons, J., Thimble, S. W., & Paine, L. K. (2000). GRASS VERSUS TREES: MANAGING RIPARIAN AREAS TO BENEFIT STREAMS OF CENTRAL NORTH AMERICA.

Journal of the American Water Resources Association, 36(4), 919–930. https://doi.org/10.1111/j.1752-1688.2000.tb04317.x

Lyons, J., Zorn, T., Stewart, J., Seelbach, P., Wehrly, K., & Wang, L. (2009). Defining and Characterizing Coolwater Streams and Their Fish Assemblages in Michigan and Wisconsin, USA. North American Journal of Fisheries Management - NORTH AM J FISH MANAGE, 29, 1130–1151. https://doi.org/10.1577/M08-118.1

Merriam, E. R., & Petty, J. T. (2019). Stream channel restoration increases climate resiliency in a thermally vulnerable Appalachian river. Restoration Ecology, 27(6), 1420–1428. https://doi.org/10.1111/rec.12980

Michie, R. (2022), heatsource-9. GitHub repository. https://github.com/DEQrmichie/heatsource-9

Mitro, M. G., Lyons, J. D., Stewart, J. S., Cunningham, P. K., & Griffin, J. D. T. (2019). Projected changes in Brook Trout and Brown Trout distribution in Wisconsin streams in the mid-twenty-first century in response to climate change. Hydrobiologia, 840(1), 215–226. https://doi.org/10.1007/s10750-019-04020-3

Moore, D. R.., Spittlehouse, D. L., & Story, A. (2005). Riparian Microclimate and Stream Temperature Response to Forest Harvesting: A Review1. JAWRA Journal of the

American Water Resources Association, 41(4), 813–834. https://doi.org/10.1111/j.1752-1688.2005.tb03772.x

Murdock, E. (2017). Understanding local climate change impacts on water resources in southern Wisconsin, and the implications for managed intensive grazing. (Dissertation, University of Wisconsin-Madison) .

O'Briain, R., Shephard, S., & Coghlan, B. (2017). River reaches with impaired riparian tree cover and channel morphology have reduced thermal resilience. Ecohydrology, 10(8), e1890. https://doi.org/10.1002/eco.1890

Olson, K. (2022). Evaluation Of Trout Population Trends And Fisheries Management In The Bear Creek Watershed, Vernon And Monroe Counties, WI. Wisconsin Department of Natural Resources - Bureau of Fisheries Management.

Olson, K. (2023). West Fork Kickapoo River Creel Survey. Trout Unlimited Driftless Science Symposium (virtual). March 7, 2023. https://www.youtube.com/watch?v=DIv2dwIlXRI

Olson, K., Mauel, K., & Pechacek, K. (2021). Evaluation of trout population trends and fisheries management in the West Fork Kickapoo River Watershed. Wisconsin Department of Natural Resources - Bureau of Fisheries Management.

Oregon Department of Environmental Quality (Oregon DEQ) (2006). Willamette Basin Total Maximum Daily Load (TMDL) Appendix C: Temperature. https://www.oregon.gov/deq/FilterDocs/appxctemp.pdf

Ouellet, V., St-Hilaire, A., Dugdale, S. J., Hannah, D. M., Krause, S., & Proulx-Ouellet, S. (2020). River temperature research and practice: Recent challenges and emerging opportunities for managing thermal habitat conditions in stream ecosystems. Science of The Total Environment, 736, 139679. https://doi.org/10.1016/j.scitotenv.2020.139679

Parent, J. R. (n.d.). Using Leaf-off LiDAR in Modeling Forest Canopy Structure and Assessing the Effect of Spatial Resolution in Landscape Analyses.

Petty, J. T., Hansbarger, J. L., Huntsman, B. M., & Mazik, P. M. (2012). Brook Trout Movement in Response to Temperature, Flow, and Thermal Refugia within a Complex Appalachian Riverscape. Transactions of the American Fisheries Society, 141(4), 1060–1073. https://doi.org/10.1080/00028487.2012.681102

Planet Team (2017). Planet Application Program Interface: In Space for Life on Earth. San Francisco, CA https://api.planet.com/

Plowright, Andrew (2024). Forest Tools. https://github.com/andrew-plowright/ForestTools.

Poole, G. C., & Berman, C. H. (2001). An Ecological Perspective on In-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-CausedThermal Degradation. Environmental Management, 27(6), 787–802. https://doi.org/10.1007/s002670010188

Potter, K.W. (2019). Hydrology of the Driftless Area. Driftless Science Symposium.

Ross, J. (2013). Trout Unlimited's Guide to America's 100 Best Trout Streams, Updated and Revised. Rowman & Littlefield.

Schuster, Zachary. (2017). Production of Decision Relevant Science for Understanding the Spatial Distribution of Baseflow and Stream Temperatures in Driftless Area Headwaters. (Dissertation, University of Wisconsin - Madison)

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Soil Survey Geographic (SSURGO) Database. at https://sdmdataaccess.sc.egov.usda.gov.

Spanjer, A. R., Gendaszek, A. S., Wulfkuhle, E. J., Black, R. W., & Jaeger, K. L. (2022). Assessing climate change impacts on Pacific salmon and trout using bioenergetics and spatiotemporal explicit river temperature predictions under varying riparian conditions. PLOS ONE, 17(5), e0266871. https://doi.org/10.1371/journal.pone.0266871

Sparrow, O. (2018). Analysis of Shade Provided by Grassy and Woody Riparian Vegetation along a Small Coldwater Stream in the Upper Midwestern United States (Master's thesis, University of Minnesota).

Stamp, J., Hamilton, A. I., Craddock, M., Parker, L., Roy, A. H., Isaak, D. J., ... & Bierwagen, B. (2014). Best practices for continuous monitoring of temperature and flow in wadeable streams (No. EPA/600/R-13/170F). US Environmental Protection Agency.

Stewart, J. S., Westenbroek, S. M., Mitro, M. G., Lyons, J. D., Kammel, L. E., & Buchwald, C. A. (2015). A model for evaluating stream temperature response to climate change in Wisconsin (No. 2014-5186). US Geological Survey.

Trimble, S. W. (1997). Stream channel erosion and change resulting from riparian forests. Geology, 25(5), 467. https://doi.org/10.1130/0091-7613(1997)025<0467:SCEACR>2.3.CO;2

Trimble, S. W. (1999). Decreased Rates of Alluvial Sediment Storage in the Coon Creek Basin, Wisconsin, 1975-93. Science, 285(5431), 1244–1246. https://doi.org/10.1126/science.285.5431.1244

Trimmel, H., Gangneux, C., Kalny, G., & Weihs, P. (2016). Application of the model ‘Heat Source’to assess the influence of meteorological components on stream temperature and simulation accuracy under heat wave conditions. Meteorolog Z, 25, 389-406.

Trout Unlimited (2023). Driftless Area Restoration Effort. Trout Unlimited. https://www.tu.org/conservation/conservation-areas/watershed-restoration/driftless-area-restoration-effort/

Trout Unlimited. (n.d.). Driftless Area Restoration Effort. Retrieved from https://wicouncil.tu.org/tu-projects/driftless-area-restoration-effort

Van Der Loo, M. van der. (2024). Markvanderloo/simputation [R]. https://github.com/markvanderloo/simputation (Original work published 2016)

van Vliet, M. T. H., Franssen, W. H. P., Yearsley, J. R., Ludwig, F., Haddeland, I., Lettenmaier, D. P., & Kabat, P. (2013). Global river discharge and water temperature under climate change. Global Environmental Change, 23(2), 450–464. https://doi.org/10.1016/j.gloenvcha.2012.11.002

Vetrano, D. M. (2019). History of Restoration: Destruction, Renewal, and Hope for the Future of Driftless Area Trout Streams. Driftless Science Symposium.

Washington DNR (n.d.). Focus—Effects of Elevated Water Temperatures on Salmonids. https://apps.ecology.wa.gov/publications/publications/0010046.pdf

Webb, B. W., & Zhang, Y. (1997). Spatial and seasonal variability in the components of the river heat budget. Hydrological processes, 11(1), 79-101.

Wehrly, K. E., Wang, L., & Mitro, M. (2007). Field-Based Estimates of Thermal Tolerance Limits for Trout: Incorporating Exposure Time and Temperature Fluctuation. Transactions of the American Fisheries Society, 136(2), 365–374. https://doi.org/10.1577/T06-163.1

Wisconsin Department of Administration - Geographic Information Office. (2020). LiDAR-Derived Countywide 60-cm Bare Earth DEM for Vernon County, WI 2020. https://geodata.wisc.edu/catalog/4df8c424-0911-400f-8b03-313344b0de56

Wisconsin DNR (2002). Conservation Easements: A landowners guide to easements purchased through WI's Runoff Management and Stewardship Streambank Protection Program. https://embed.widencdn.net/pdf/plus/widnr/fqjiabrdyd/PUB_WT_0549_2002.pdf?u=6c5niw

Wisconsin DNR (2023). Expenditures of Inland Trout Stamp Revenues, 2019-2021. https://widnr.widen.net/s/xxhd8psbrw/expendituresinlandwatertroutstamprevenues20192021

Wisconsin Initiative on Climate Change Impacts (WICCI). (2011). Wisconsin's Changing Climate: Impacts and Adaptation.

Wisconsin Regional Orthoimagery Consortium. (2020). WROC Aerial Mosaic (WTM) Vernon County, WI 2020. https://geodata.wisc.edu/catalog/a32841cb-e06c-402f-81f6-47f5ce73a1b6

Wood, J. R. (n.d.). Stream Temperature Monitoring and Modeling to Inform Restoration: A Study of Thermal Variability in the Western US.

Appendix A

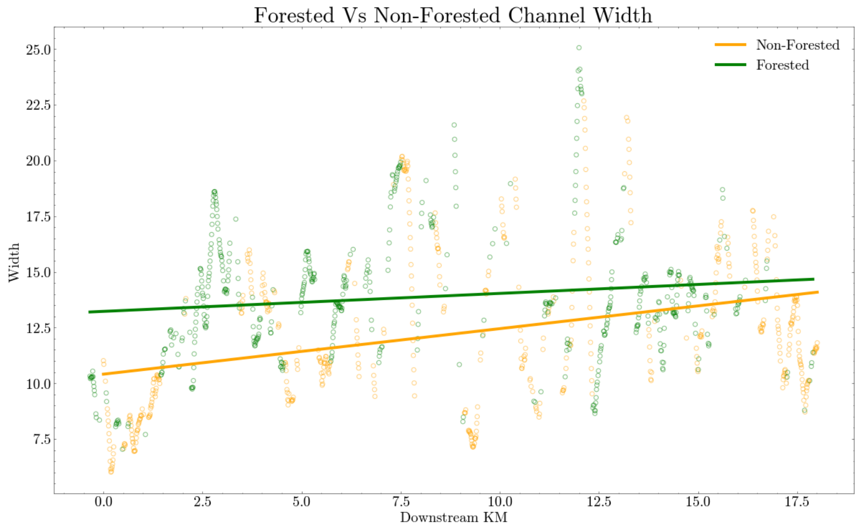

Figure A1: Forested and non-forested channel width.

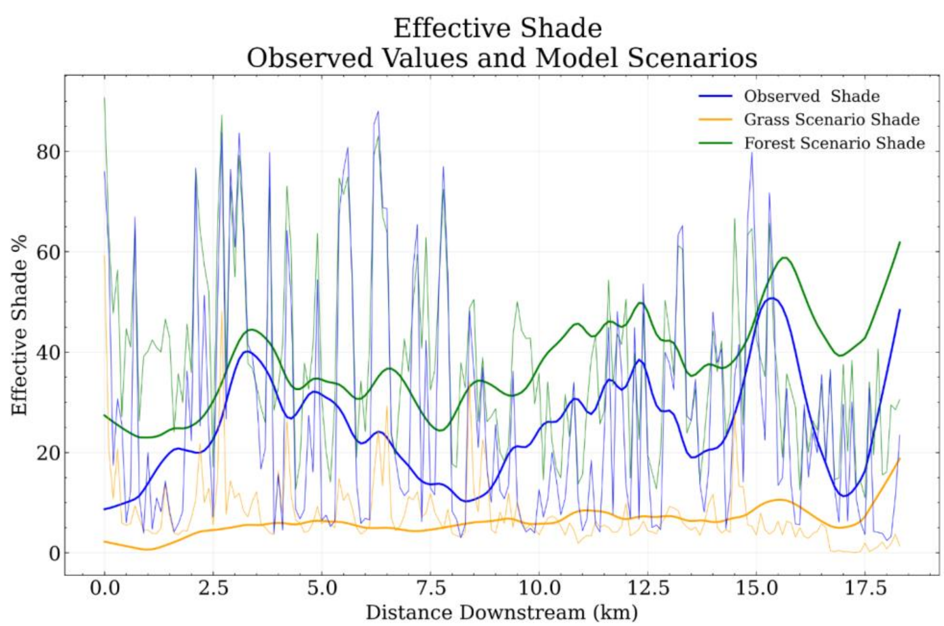

Figure A2: Trends in effective shade - observed and modeled baseline conditions.

Lidar Crown Max height < 5 ft = 1.76 m
Lidar Crown Max height <10 2 = 2.28 m
Lidar Crown Max height <15 ft = 3.38 m
Lidar Crown Max height <20 ft = 5.34 m
Lidar Crown Max height<25 ft = 6.85 m
Lidar Crown Max height <30 ft = 8.38 m
Lidar Crown Max height <40 ft = 10.67 m
Lidar Crown Max height <50 ft = 13.71 m
Lidar Crown Max height <60 ft = 16.74 m
Lidar Crown Max height <80 ft = 21.33 m
Lidar Crown Max height <100 = 27.43 m

Figure A3: LiDAR tree canopy classifications for TTools.

```
WHEN "geomdesc" = 'depressions on flood plains, overflow stream
channels on flood plains' THEN 1
WHEN "geomdesc" = 'depressions on flood plains on river valleys, swales
on flood plains on river valleys' THEN 1
WHEN  "geomdesc" = 'drainageways' THEN 1
WHEN "geomdesc" = 'drainageways on hills, alluvial fans on hills' THEN 1
WHEN  "geomdesc"  = 'drainageways on stream terraces' THEN 1
WHEN  "geomdesc"  = 'flats on flood plains' THEN 1
WHEN  "geomdesc"  = 'flood plains' THEN 1
WHEN  "geomdesc"  = 'hills' THEN 3
WHEN  "geomdesc"  = 'hills, uplands' THEN 3
WHEN  "geomdesc" = 'interfluves on uplands' THEN 3
WHEN  "geomdesc" = 'knolls on river valleys' THEN 2
WHEN  "geomdesc" = 'natural levees on stream terraces' THEN 1
WHEN  "geomdesc" = 'pediments' THEN 1
WHEN  "geomdesc" = 'ridges on uplands' THEN 3
WHEN  "geomdesc" = 'river valleys, valley sides' THEN 2
WHEN  "geomdesc" = 'sand sheets on hills' THEN 3
WHEN  "geomdesc" = 'stream terraces' THEN 1
WHEN "geomdesc"  = 'uplands, valley sides' THEN 2
WHEN  "geomdesc" = 'valley sides on uplands' THEN 2
WHEN  "geomdesc" = '-- Error in Exists On --' THEN 1
```

Figure A4: QGIS code to classify SSURGO physiographic groupings into hydrology model inputs. (class 1=valley, class 2 = hillslopes, class 3 = ridgetops)

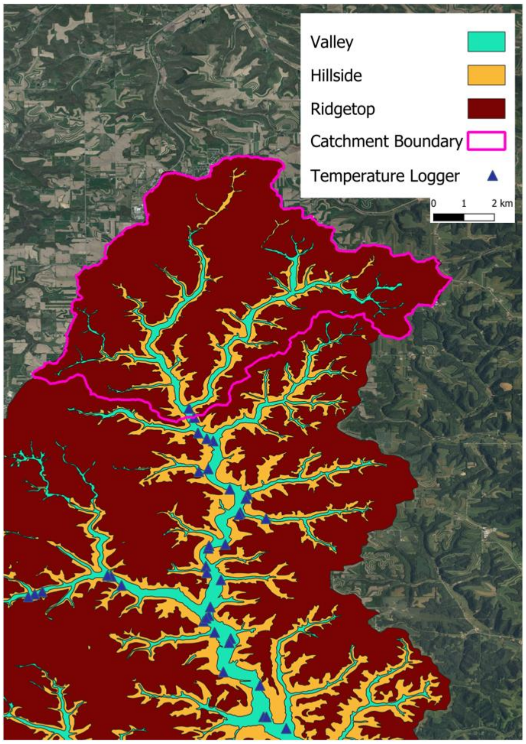

Figure A5: Example determination of physiographic type in a contributing area.

Commented [BS115]: Could go in supplementary

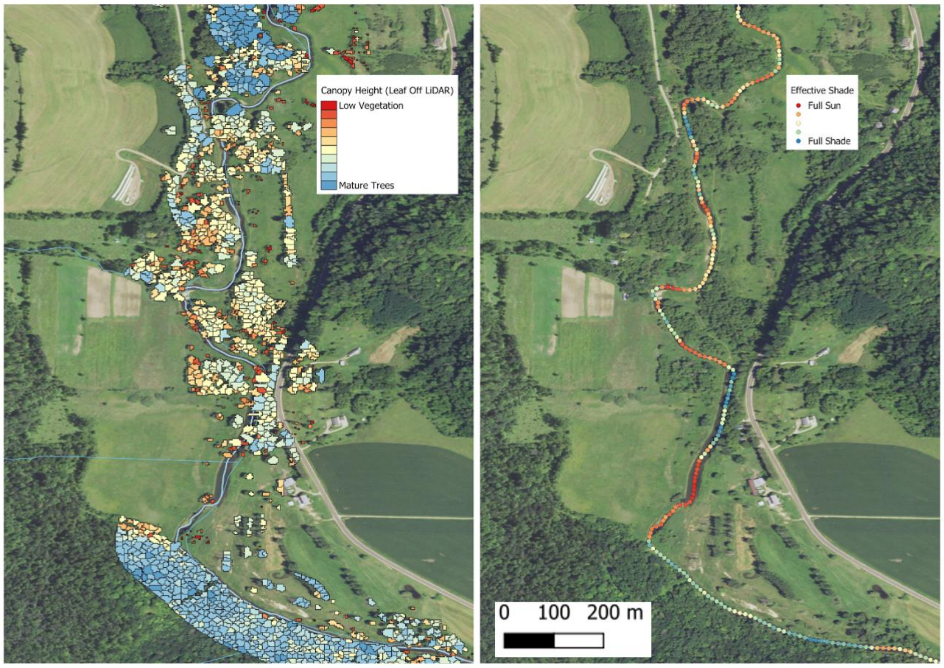

Figure A6: Example of LiDAR Canopy polygons and associated shade calculations.

Additional management context:

The Driftless Area in the Southwest corner of Wisconsin is home to stream-dwelling salmonid populations and abundant cold-water resources. Surface water in the Driftless Area is dominated by spring fed streams, renowned for their biologic richness and cold temperatures year-round. Recreation is a high priority in watersheds that host game fish, namely native Brook Trout (*Salvelinus fontinalis*) and introduced Brown Trout (*Salmo trutta)*, which have been able to thrive in recent decades despite (and sometimes because of) large scale ecological changes brought by European settlement. The Driftless Area is a highly regarded trout fishing destination among anglers, driving a culturally and economically important tourism economy (Ross, 2013)**.** Hundreds of miles of state managed fisheries areas can be found in Wisconsin's Driftless Area, with fish densities rivaling famous western fly-fishing destinations. However, temperature rise due to climate change poses a threat to the cold-water resources of the region and the fish that depend on them for each of their life stages (Olson, 2021: Elliot, 1994).

Stream restoration in the Driftless Area has primarily been intent on enhancing trout habitat in streams for the development of a recreational fishery, and to stabilize property against the erosive impacts of increasingly large floods (Trout Unlimited, 2023;). Many stream channels which were wide and slow moving in the early to mid-20th century, have been narrowed, and fish habitat for adult brown trout has been supplemented by providing overhead cover and spawning gravel through the installation of in-stream habitat structures and the mechanical slope reduction of steep banks (Vetrano, 2019). In-stream habitat restoration efforts in the Driftless Area, combined with farm scale conservation practices which improved groundwater recharge and restored baseflow, has led to recognition as a nationally renowned recreational fishery, driving recreational tourism which bolsters local economies by generating an estimated $1+ billion dollars annually (Anderson, 2016)**.** Many stream restoration projects in the region involve replacement of riparian trees, mainly boxelder (*Acer Negundo*), with cool season grass mixtures to reduce local erosion and improve angler access (Wisconsin DNR, 2023). The effect of this restoration style on stream temperature has been a topic of intense debate.

Additional site information:

The hydrology of this watershed is typical of the area, known for high base-flows relative to the rest of the state (Gebert et al., 2011). This high baseflow is partly a result of high recharge rates on surrounding hillslopes, and local hydro stratigraphy which consists of alternating layers of highly porous (sandstone, dolomite) and impermeable bedrock layers (shale). Porous bedrock strata facilitate infiltration, while impermeable bedrock creates underground flow paths, which promote groundwater discharge at surficial springs and directly to stream channels through accretion (Potter, 2019). Bedrock heterogeneity causes a large amount of groundwater discharge variability in the watershed, resulting in a non-linear pattern of stream warming moving downstream.

The WFK watershed is widely known for its value as a recreational trout fishery, one of the most popular trout fishing destinations in the state for anglers as shown by recent Wisconsin DNR creel surveys (Olson, 2023). The watershed contains ~109 km of classified trout streams, of which 25 km are under easements for public access and managed by Vernon County and the Wisconsin DNR.

Additional Temperature Logger information:

Temperature monitoring sites were chosen based on their depth and flow characteristics, prioritizing channel cross sections with a maximum depth of 0.5-1 meters and well mixed flow conditions (Stamp et al., 2014). Care was taken to avoid placing loggers in areas of slow velocity where temperature stratification could exist. Temperature loggers were placed 5-10 cm above the bed surface to avoid being covered by mobile sediment while also limiting the effect of direct solar radiation on water temperature measurements.

All permanent temperature sensors were placed in channels residing on state managed easements, or on private land with landowners' prior approval. In areas without easements, like small tributaries and areas where landowners were unreachable, temporary water temperature sensors were deployed for 1-2 days anchored to metal railroad plates and placed on the stream bed. Temporary sensor sites were left in the stream for at least one complete diurnal cycle, and in some cases multiple days throughout the summer.

Measurements of shallow groundwater temperature were collected using a water temperature logger ~2.5 m below the ground surface, in a water table well 10m away from the stream channel near the downstream boundary of the study area. A HOBO U20-001-04 water level logger was used to collect shallow groundwater temperature. Water table elevation was ~1.5 m below the ground surface. The well was dug deep enough to ensure the sensor was submerged throughout the field season at a depth of ~0.5 m.

Additional Meteorological Data Information

Hourly observations of solar radiation from a continuously logging pyranometer attached to our weather station were used to calculate the ‘cloudiness index’ for use in the Heat Source solar radiation module. This index relies on observations of incoming solar radiation and calculations of potential incoming solar radiation for a given latitude, longitude, and elevation. The cloudiness index is the proportion of observed solar radiation to the potential solar radiation subtracted from 1 to get a decimal fraction.

$$Cloudiness\ Index = 1 - \frac{Observed\ Solar\ Radiation}{Potential\ Solar\ Radiation} \qquad \text{(Equation 1A)}$$

Additional Geomorphology Input Information:

Shapefiles resulting from digitization of left bank, right bank, and stream centerline were used in the Heat Source 9's companion ArcGIS plugin scripts, TTools. Widths are determined by sampling the distance between left and right bank shapefiles at each model node location. Because of errors caused by local variation in channel width at 100m node spacing, model nodes were assigned an average of channel width measured at 10m intervals within the adjacent 50m upstream and downstream. Results of width measurements were validated with observed width measurements collected in the field. Model node elevations were sampled in GIS using 10-meter resolution LiDAR digital elevation.

Additional Streamflow and Groundwater Discharge Information

Landform classification shapefiles were synthesized from USDA Soil Survey Geographic Database (SSURGO) geomorphic classifications (Soil Survey Staff). Criteria used to group SSURGO geomorphic regions into valley, hillslope, and ridgetop can be seen in Figure A4. Small adjustments were made to ensure the continuity of hillslopes as well as to assure polygons were not classified as ridgetops when they are at lower elevations than adjacent hillslopes.

To determine the accretion rate between modeling nodes, a python script was developed to automate the delineation of watershed contributing areas for nodes spaced at 100m moving down the stream centerline. Watershed delineation used a 10m LiDAR DEM, automated using the PySheds python package (Bartos, 2020). For each contributing area polygon, the area of valley, hillside, and ridgetop classes were extracted by clipping the larger SSURGO landforms shapefile to the contributing area polygon (Figure 3). Areas of each landform were then converted to base-flow discharge using values from Juckem (2006) (Table 1). Model accretion values were determined by subtracting upstream baseflow estimates from downstream estimates to represent the amount of groundwater discharge in the 100m reach.

Appendix B

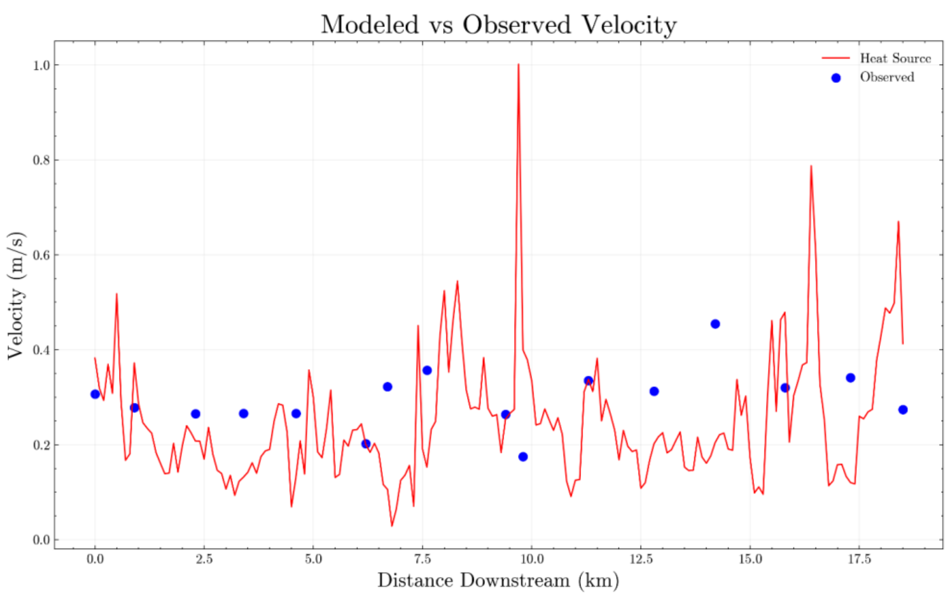

Figure B1: Modeled and observed flow velocity.

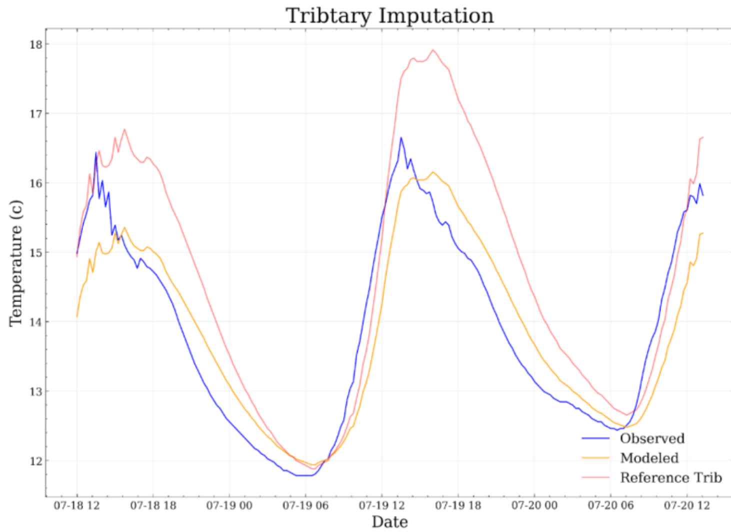

Figure B2: Comparison of tributary (7.7 km) to nearby tributary used for modeling and observed temperatures.

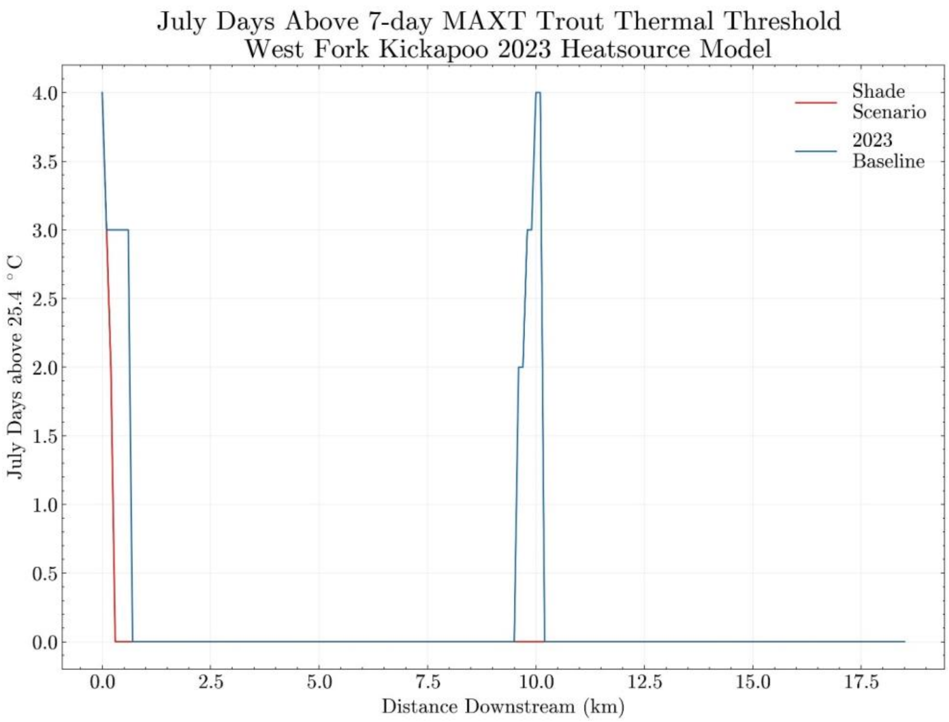

Figure B3: Reduction of days above the 7-D MAXT thermal threshold for trout .

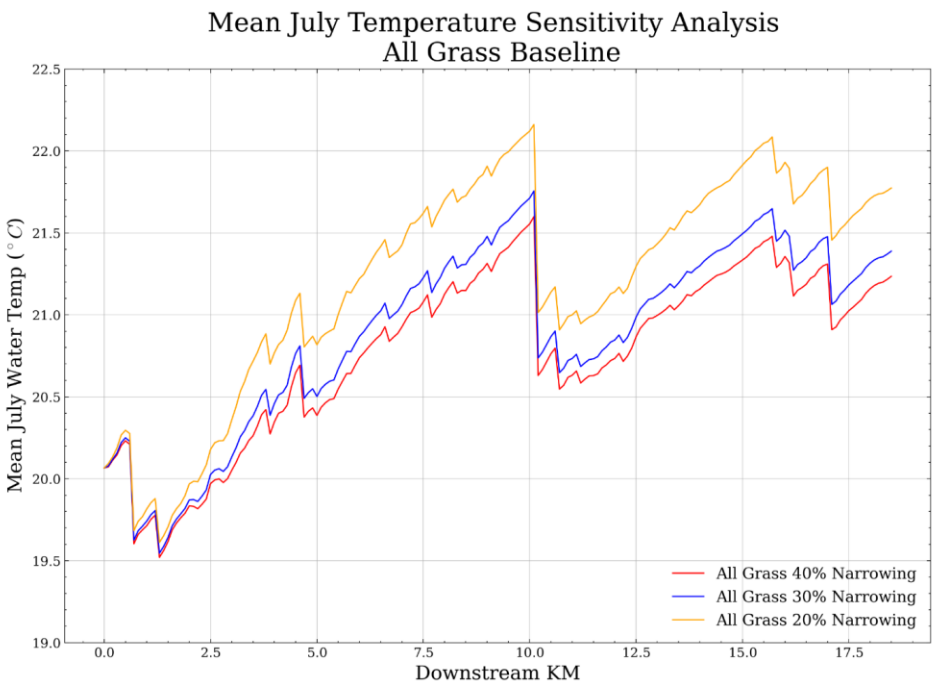

Figure B4: Sensitivity of mean July water temperature to channel narrowing. Uniform grassed riparian zone.

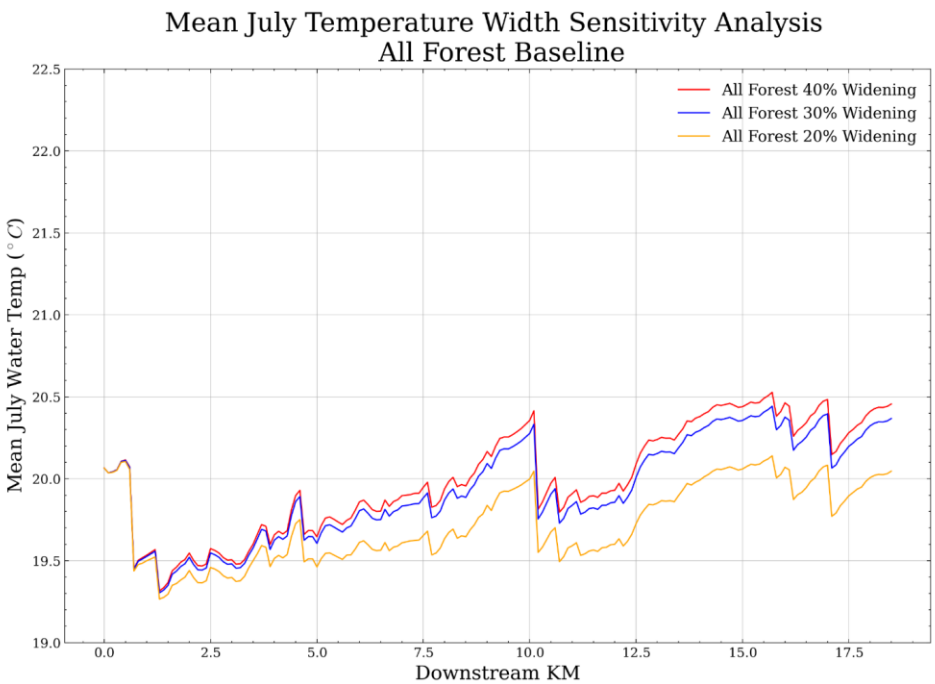

Figure B5: Sensitivity of mean July water temperature to channel widening. Uniformly forested riparian zone.

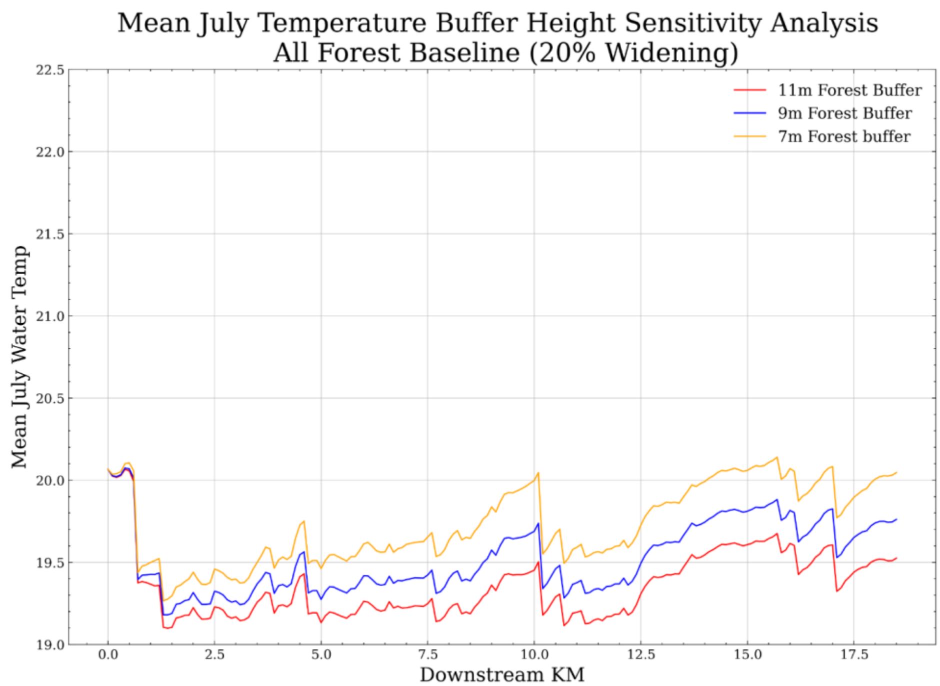

Figure B6: Sensitivity of mean July water temperature to forest buffer heights.

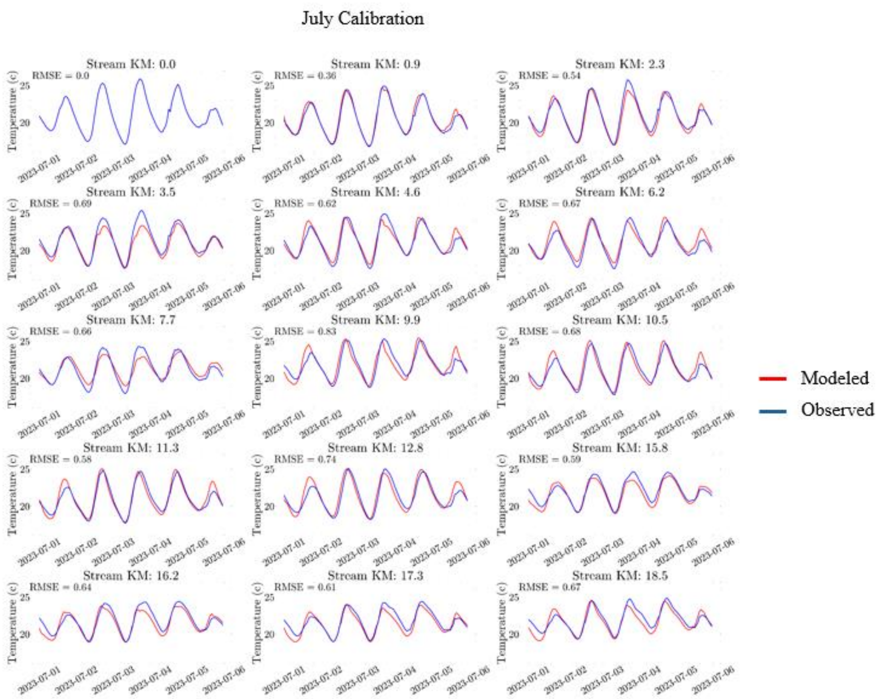

Figure B7: July model compared to observed temperatures.

Commented [EB119]: You could probably put this figure in the Appendix

Commented [BS120]: Might be too many sites, pick ~3 that are spaced throughout the reach

Commented [BS121]: Update title because it is more of a comparison between modeled and observed temps during a shorter model period.

Baseflow Discharge Modeling Results

The results of discharge modeling using the proportions of physiographic regions (valley, hillslope, and ridgetop) in mainstem and tributary WFK sites are presented below (Table 5, Table 6, Figure 9, Figure 10)**.** The predictions of baseflow discharge were compared to observed discharge measurements collected in the field during baseflow conditions throughout the summer of 2023. Overall root mean square error (RMSE) was .07 m^3/s in mainstem sites and .03 m^3/s in tributary sites. The largest errors between modeled and observed values are seen at main stem station 11.1, this error may stem from erroneous values observed during field measurements. Modeled discharge values slightly underestimate discharge at the upstream and downstream end of the study reach, while potentially overestimating values within the middle of the reach.. Future modeling could benefit from incorporating more detailed hydrogeologic data or adjusting parameterizations to better capture the variability of baseflow discharge behavior across the study area. Despite these limitations, the current model serves as a valuable tool for understanding the broader trends in groundwater discharge within the region and serves as a tool in assessing the influence of groundwater accretion on stream temperature dynamics between model nodes where measurements of groundwater discharge were not feasible.

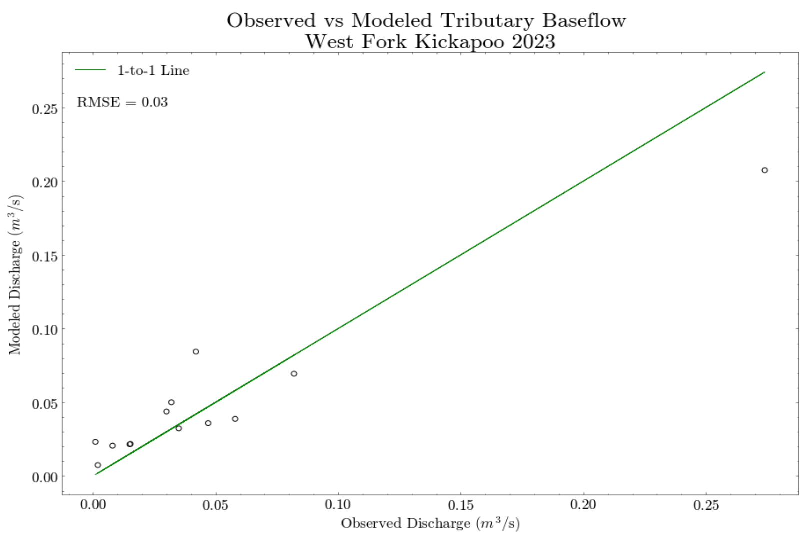

Figure B8:Comparison of Modeled and observed tributary discharge.

Station Downstream Distance	Valley Area (km^2)	Hillside Area (km^2)	Ridgetop Area (km^2)	Total Area (km^2)	Observed Discharge (m^3/s)	Modeled Discharge (m^3/s)	Model Error (m^3/s)
0.70	0.40	0.77	5.79	6.96	0.058	0.039	-0.019
1.20	1.27	3.06	8.97	13.29	0.042	0.084	0.042
2.60	0.18	0.80	2.79	3.76	0.001	0.023	0.022
3.90	0.48	1.73	5.85	8.06	0.032	0.050	0.018
4.70	0.43	1.30	5.59	7.32	0.030	0.044	0.014
6.70	0.16	0.53	3.07	3.76	0.015	0.022	0.007
7.80	0.08	0.25	0.88	1.21	0.002	0.007	0.005
8.30	0.12	0.73	2.48	3.33	0.008	0.021	0.013
10.20	3.05	4.98	27.88	35.91	0.274	0.207	-0.067
10.70	0.31	0.97	4.13	5.40	0.035	0.032	-0.003
15.80	0.51	1.40	3.62	5.53	0.047	0.036	-0.011
16.10	0.26	1.23	1.40	2.89	0.015	0.022	0.006
17.00	0.80	3.01	6.57	10.38	0.082	0.069	-0.013

Table B1: Results of tributary discharge modeling.

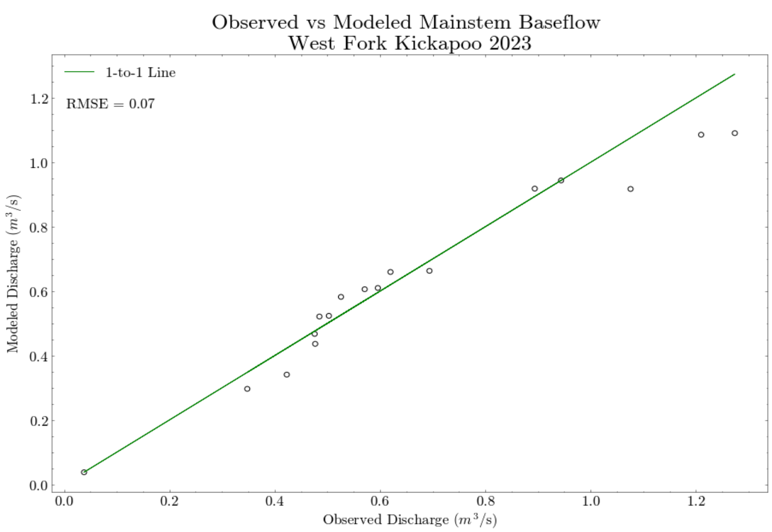

Figure B9: Comparison of Modeled and observed mainstem discharges.

Station Downstream Distance	Valley Area (km^2)	Hillside Area (km^2)	Ridgetop Area (km^2)	Total Area (km^2)	Observed Discharge (m^3/s)	Modeled Discharge (m^3/s)	Model Error (m^3/s)
0.0	3.66	6.38	42.48	52.52	0.348	0.297	-0.051
0.9	4.24	7.42	48.56	60.22	0.423	0.341	-0.082
2.3	5.84	11.07	58.05	74.96	0.483	0.436	-0.047
3.5	6.23	12.30	61.19	79.72	0.476	0.467	-0.009
3.9	6.87	14.23	67.20	88.30	0.485	0.521	0.036
4.6	7.01	14.34	67.22	88.57	0.503	0.523	0.020
6.2	7.84	16.45	73.58	97.87	0.526	0.582	0.056
6.7	8.14	17.10	76.66	101.90	0.571	0.606	0.035
7.6	8.26	17.36	76.73	102.35	0.596	0.610	0.014
9.4	9.01	19.20	81.89	110.11	0.620	0.660	0.040
9.9	9.28	19.34	81.96	110.58	0.694	0.663	-0.031
11.3	13.08	25.95	114.88	153.91	1.076	0.917	-0.159
12.8	13.12	26.01	114.89	154.03	0.894	0.918	0.024
13.3	13.63	26.68	115.74	156.05	0.907	0.933	0.026
14.1	14.06	27.23	116.20	157.48	0.944	0.944	0.000
15.8	14.45	27.68	116.30	158.43	1.078	0.952	-0.126
17.3	16.45	33.66	128.02	178.13	1.210	1.086	-0.124
18.5	16.70	33.90	128.15	178.75	1.274	1.091	-0.183

Table B2: Mainstem discharge model results.

Additional results of shade modeling

Using the results of modeled effective shade from county LiDAR collected in the spring of 2020, shade estimates using vegetation heights can be seen in Figure 11. The outputs show trends in shade where topography interacts with or modeled effective shade capture contributions of shade from topography and vegetation.

Additional Vegetation Measurement Information

Crown delineations shapefiles were assigned the height of the maximum height value within the 1.5m bin in which they fell. Within GIS the resulting canopy polygons were converted to a land cover raster for use in sampling vegetation shading at model nodes. Canopy polygon rasters were assigned land cover codes 1-11 based on their heights from 0 to 30 meters (Appendix 3). The resulting canopy height rasters are sampled using the TTools ArcGIS package provided with Heat Source and used to calculate shade contributed to the channel by vegetation. Heat Source contains methods for calculating the effective shade at a model node using data sampled from tree height polygons. Eight transects radiating from the model node were used to sample land cover polygons at 5 meter spacing, for use in shade calculations. At each model node, incoming shortwave radiation is attenuated by vegetation cover adjacent to the node. Heat Source shade modeling routines determine the amount of incoming solar radiation blocked by riparian vegetation as a function of vegetation height, distance from the model node, and the density of vegetation present. These variables are used to determine the view to sky for each model node, which is used in Heat Source to calculate longwave and shortwave radiation fluxes at the model node (Kalny et al., 2017). Vegetation canopy closure was used as a calibration parameter throughout the model calibration process, bounded by values used by similar studies using the Heat Source temperature model (Bond et al., 2015).

Temperature Observation Results

Table 3 summarizes observations of hourly water temperature throughout the study during the period of May-August 2023. During this period, the warmest monthly mean temperatures were observed at the downstream boundary site, within a range between 16.37℃ in

May to 20.61℃ in August. The lowest monthly means were observed at site 0.8km during May, June, July and August with a range from 15.27 ℃ in May to 19.5 ℃ in August. Monthly maximum temperatures peaked at the 9.8 km site downstream from the model boundary. Temperature records were not collected during May at this site, and are incomplete in June, however, maximum temperatures in July and August ranged from 26.30 ℃ to 26.93 ℃ respectively. The maximum temperature throughout all summer months was 27.456 ℃, observed at the upstream boundary site on July 27th, 2023, at 5:00 pm. The lowest monthly maximum temperature recorded at all mainstream temperature logger sites was on May 30th, 2023, at 5:00pm at 17.1 km.

	STREAM KM	MEAN TEMP (°C)	Δ MEAN (°C)	MAX TEMP (°C)	Δ MAX (°C)	MAX 7-D MEANT (°C)	MAX 7-D MAXT (°C)
May	0	15.64		23.81		16.46	22.07
	0.8	15.27	-0.36	22.13	-1.68	15.97	20.56
	2.2	15.57	0.30	22.51	0.38	16.43	20.92
	7.6	15.72	0.15	21.60	-0.91	16.74	20.03
	11.1	15.86	0.14	22.13	0.53	16.87	20.62
	13.9	16.10	0.24	21.80	-0.33	17.21	20.35
	17.1	16.22	0.12	21.29	-0.50	17.36	19.90
	18.3	16.37	0.15	21.60	0.31	17.54	20.23
June	0	19.44		26.65		21.16	25.21
	0.8	18.89	-0.55	25.45	-1.19	20.56	24.41
	2.2	19.25	0.37	25.60	0.15	20.85	24.72
	6.1	19.08	-0.17	24.82	-0.78	20.59	23.72
	7.6	19.34	0.26	24.65	-0.17	20.85	23.58
	11.1	19.22	-0.12	25.14	0.48	20.60	23.67
	17.1	19.75	0.52	24.41	-0.73	21.18	23.40
	18.3	19.97	0.22	25.04	0.63	21.42	23.76
July	0	20.07	0.00	27.46	0.00	21.66	25.69
	0.8	19.46	-0.61	26.06	-1.40	20.99	24.48
	2.2	19.95	0.49	26.48	0.41	21.47	24.81
	3.4	20.31	0.36	26.35	-0.12	21.84	24.66
	4.5	20.10	-0.20	26.06	-0.29	21.62	24.62
	6.1	19.87	-0.24	25.72	-0.34	21.38	24.26
	7.6	20.18	0.32	25.79	0.07	21.71	24.24
	9.8	20.84	0.65	26.74	0.95	22.41	25.13
	10.3	20.10	-0.74	26.11	-0.64	21.56	24.52
	11.1	20.05	-0.04	26.11	0.00	21.52	24.60
	12.6	20.36	0.31	26.45	0.34	21.88	24.87
	15.6	20.94	0.58	25.87	-0.59	22.52	24.45
	16	20.77	-0.17	25.70	-0.17	22.34	24.27
	17.1	20.72	-0.05	25.74	0.05	22.29	24.29
	18.3	20.98	0.26	26.21	0.46	22.60	24.72
August	0	20.08		27.24		21.46	25.17
	0.8	19.50	-0.58	26.13	-1.10	20.90	24.11
	3.4	20.10	0.60	26.18	0.05	21.75	24.37
	6.1	19.68	-0.42	26.06	-0.12	21.37	23.96
	9.8	20.54	0.86	27.48	1.42	22.42	25.06
	10.3	19.76	-0.78	26.40	-1.08	21.39	24.17
	11.1	19.72	-0.04	26.45	0.05	21.34	24.18
	15.6	20.57	0.85	26.26	-0.20	22.37	24.41
	16	20.43	-0.14	26.21	-0.05	22.21	24.27
	17.1	20.38	-0.05	26.33	0.12	22.17	24.17
	18.3	20.61	0.23	26.77	0.44	22.42	24.47

Table B3: Observed stream temperature summaries for the summer of 2023.

The difference in mean stream temperature between a site and its closest upstream node is represented by delta mean. Negative values of delta mean downstream reduction of mean water temperatures. The maximum amount of cooling between two nodes was a .77 ℃ decrease in mean temperature between sites at 9.8km and 10.3km. This large decrease likely reflects the addition of the largest tributary, Seas Branch, between the two nodes. The highest increase in monthly mean temperatures between two nodes was observed between sites at 11.1km and 15.6 km, with an increase of .85 ℃ in mean monthly temperature.

Delta max represents the change in maximum water temperature between nodes downstream. The largest decrease in maximum temperature, like mean temperature, came between sites 9.8 and 10.3 with a difference of -1.079 ℃ in August maximum temperature. The largest increase in maximum monthly temperature was observed between the site at 6.1 km and 9.8 km with an increase of 1.42 ℃ in August and 1.24 ℃ in July.

Gap Filling Tributary Temperatures Results

Results of tributary gap filling using nearby continuous tributary data showed a moderate ability to predict diurnal temperature trends when no sensor data was available (Table 4). RMSE of observed and simulated tributary temperature values range from .2 to 1.36 ℃. Multiple weeks of data were recorded at model km 10.5 and 15.7. The higher error in these sites in comparison to others may be a result of having a longer temperature record than other sites. A comparison between observed and modeled tributary data at site 7.7 can be seen in Appendix 5. The density of small tributaries within our study site was a challenge and limitation of this study, and gap filling allowed us to extend the record of small tributary temperatures within a realistic range using statistical relationships.

	2.5 km	3.8 km	4.6 km	7.7 km	8.2 km	10.0 km	10.5 km	15.7 km	16.0 km
RMSE	0.55	0.40	0.23	0.59	0.54	0.20	1.04	1.36	0.61
MSE	0.30	0.16	0.05	0.35	0.30	0.04	1.08	1.84	0.37
MAE	0.48	0.31	0.18	0.50	0.44	0.16	0.88	1.10	0.47
NSE	0.61	0.97	0.98	0.82	0.94	0.99	0.73	0.86	0.93
R^2	0.61	0.97	0.98	0.82	0.95	0.99	0.73	0.86	0.95
Metric	0.55	0.40	0.23	0.59	0.54	0.20	1.04	1.36	0.61
RMSE	0.30	0.16	0.05	0.35	0.30	0.04	1.08	1.84	0.37

Table B4: Tributary gap filling error metrics.