# Influences of riparian vegetation on trout stream temperatures in the North Central Hardwoods Forest Ecoregion of Wisconsin

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# ABSTRACT

Brook trout distribution and abundance throughout Wisconsin has been reduced compared to historical levels. Much of this reduction is believed to be due to land use changes that have increased summer stream temperatures which reduce the length of stream thermally suitable for brook trout (*Salvelinus fontinalis*). Brook trout prefer temperatures ranging from 10 to 19°C, and 22.3°C is considered suitable when maximum weekly average temperatures (MWAT) are attained. Trees in the riparian area reduce stream temperatures by adding shade to the stream and altering the microclimate to promote cooler stream temperatures. Managing riparian vegetation offers potential for mitigating increased stream temperatures caused by poor land use and other factors thereby providing an opportunity to increase the amount of stream with suitable temperatures for brook trout. This study assessed the potential for forested riparian areas to create thermally suitable stream segments for brook trout in central Wisconsin's North Central Hardwoods Forest Ecoregion. Temporal thermal profiles were created for twelve streams, six in 2007 and six in 2008, throughout central Wisconsin for the MWAT time period. Five of the twelve streams monitored became thermally unsuitable for brook trout with study stream temperatures ranging from 13.3 to 23.6°C during the MWAT period. Using a stream heat budget temperature model (Stream Segment Temperature Model), stream temperatures were modeled for the MWAT period under varying levels of riparian vegetation stream shading (0, 25, 50, and 75%). Modeling results were then compared to the temporal thermal profiles in order to assess the influence of stream shading on the length of stream thermally suitable for brook trout. Shading modeled at the 0% level predicted average equilibrium stream temperatures of 25.7°C and decreased the length of thermally suitable stream for brook trout by as much as 91.5%, (4.31 km, on Unnamed 17-5 / Cunningham Creek in Clark County). On the other hand, shading

modeled at the 75% level predicted average equilibrium stream temperatures of 20.9°C and increased the suitable length of stream by as much as 128.63%, (4.96 km, on Sucker Creek in Waushara County). Multiple regression analysis found upstream temperature, segment length, stream width, and width to depth ratio to be positively related to downstream temperature, whereas change in flow per distance, stream slope, riparian vegetation shade, sand substrate, and upstream flow were inversely related. Analysis of covariance found stream reaches with riparian trees to be significantly cooler compared to grass-vegetated reaches. Forested riparian areas had stream temperatures 0.74°C colder than grass vegetated riparian areas at the downstream end of stream reaches during the MWAT period and 0.93°C colder during the maximum daily average temperature (MDAT) period. Estimated marginal means (i.e., least squared means) for change in stream temperature per km predicted a decrease of 0.54°C/km under forested riparian areas compared to an increase of 1.23°C/km under grass-vegetated riparian areas during the MDAT period. In summary riparian forests are clearly important for maintaining thermal conditions suitable for brook trout and provide an opportunity to increase the amount of thermally suitable brook trout water in central Wisconsin.

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# INTRODUCTION

Prior to early European settlement of Wisconsin's North Central Hardwood Forests Ecoregion (Omernik and Gallant 1988), the region's riparian areas were dominated by pristine hardwood forests (Curtis 1959; Verry and Dolloff 2000). During this period, many of the area streams contained abundant brook trout (*Salvelinus fontinalis*) populations (Becker 1983). However, since early settlement, land use activities including logging, grazing, crop agriculture, and urbanization have altered riparian areas and their vegetation throughout Wisconsin (Curtis 1959; Knox 1977; Wang et al. 1997). These changes contributed to increased stream temperatures forcing brook trout into upstream stream reaches to seek thermal refugia, or in some areas they were extirpated (Becker 1983). Since brook trout distribution is strongly linked to stream temperatures and stream temperatures are linked to riparian vegetation, there is an inherent linkage between trout and riparian vegetation. By quantifying the relationship between riparian vegetation and stream temperatures in central Wisconsin the opportunity to restore brook trout waters can be realized.

Temperature is an important factor influencing the distribution and abundance of trout (Binns and Eiserman 1979; Bowlby and Roff 1986; Bozek and Hubert 1992; Stoneman and Jones 2000; Wehrly et al. 2003). Brook trout can only survive across a narrow thermal range; they have a physiological requirement of cold water temperatures and high concentrations of dissolved oxygen (Threinen and Poff 1963; Cherry et al. 1977; Power 1980; Becker 1983; Countant 1987). Unsuitable water temperatures can negatively affect reproduction, disease and parasite susceptibility, predation vulnerability, interspecific competition, feeding, growth, and movement, the latter of which are effected by metabolism (Fry 1947; Brett 1956; Brett 1971;

Coutant 1973; USEPA 1976; Elliott 1981; Reeves et al. 1987; Taniguchi et al. 1998; Reese and Harvey 2002).

Brook trout studies have identified optimum temperatures for growth between 13-16.1°C (Hokanson et al. 1973; Dwyer et al. 1983) and preferred temperatures between 10-19°C (Hokanson et al. 1973; Cherry et al. 1975); the upper limit of thermal tolerance ranges from 23.5-25.5°C (Fry et al. 1946; McCormick et al. 1972; Cherry et al. 1977). Field observations have identified brook trout distributions to be limited by maximum weekly average temperatures (MWAT) in excess of 22.3°C (Eaton et al. 1995), 23.3°C (Wehrly et al. 2007), and 24°C (Meisner 1990), and limited by a maximum daily average temperatures (MDAT) in excess of 24°C (Binns and Eiserman 1979; Barton et al. 1985; Bowlby and Roff 1986; Picard et al. 2003). Brook trout are also found at much greater abundances when temperatures are closer to their preferred levels (Wehrly et al. 1998; Isaak and Huber 2004). In fact, Wehrly et al. (1998) found stream temperatures between 15-19°C had the highest densities of trout and Lyons et al. (1996) suggested coldwater fisheries should not exceed a MDAT of 22°C.

Solar radiation received by a stream is one of the most influential factors affecting stream temperatures (Brown 1970; Beschta et al. 1987; Johnson 2004). Riparian vegetation canopies promote shading thereby reducing solar radiation received by a stream leading to lower summer water temperatures and a reduction in stream temperature fluctuations (Brown 1969; Moring and Lantz 1975; Barton et al. 1985; Brown 1985; Sullivan and Adams 1991; Hetrick et al. 1998; Johnson 2004).

Thermal regimes of streams control the longitudinal distribution of brook trout because behavioral thermoregulation is the only real means for trout to cope with unsuitable water temperatures. This requires brook trout to migrate to stream reaches with more preferable temperatures or take the risk of perishing (Neil and Magnuson 1974; Beitinger and Fitzpatrick 1979; Neil 1979; Reynolds and Casterlin 1979; Meisner 1990; Kaeding 1996; Hayes et al. 1998). With many factors influencing the quantity of heat energy in a stream and discharge of a stream, the thermal regime of streams can be quite complex and stream energy budgets very dynamic (Theurer et al. 1984; Moore et al. 2005). Thermally influential factors include stream discharge, solar radiation, air temperature, windspeed, stream morphology, and groundwater inputs (Theurer et al. 1984; Sullivan and Adams 1991; Walks et al. 2000; Poole and Berman 2001; Gaffield et al. 2005). Different types and amounts of riparian vegetation can create various microclimates, influence the amount of solar radiation a stream receives, and alter groundwater inputs, which influence the thermal regime of a stream. Forest microclimates are cooler than non-forested areas in the daytime and slightly higher at nighttime, leading to lower air temperature, decreased air temperature fluctuations, and more stable and cooler water temperatures (Moore et al. 2005).

Currently, much of Wisconsin's trout stream habitat management focuses on implementing in-stream habitat structures, stabilizing banks, beaver dam removal, and streambank debrushing even though these activities have had questionable success rates (see Hunt 1988; Avery 2004). Moreover, many of these habitat management techniques require maintenance or they will degrade requiring continued upkeep costs. One of these management techniques, called bank debrushing, involves the removal of riparian shrubs and trees where shade is not needed to maintain tolerable water temperatures in hopes of increasing in-stream trout habitat and food availability (White and Bryndlson 1967; Hunt 1979). A lack of understanding what management techniques should be implemented and when shade is not needed to maintain tolerable water temperatures exists and needs to be addressed.

Management of riparian areas and streams clearly needs integration and science-based guidance so that areas where shade is needed to maintain temperatures suitable to brook trout can be maintained and streams becoming thermally marginal to brook trout can be managed to increase the length of thermally suitable stream. This study quantifies the relationship between riparian vegetation and stream temperatures in order to assess the potential for gains in thermally suitable brook trout water by managing for forested riparian areas and provides science-based models that can help guide and/or possibly change current habitat management of brook trout streams in Wisconsin. The objectives of this study are to (1) develop temporal thermal profiles for twelve trout streams in central Wisconsin to identify transitions from suitable to unsuitable thermal conditions for brook trout, (2) formulate and calibrate a stream temperature model to predict the length of stream thermally suitable to brook trout that could be gained/lost by riparian vegetation management of 0, 25, 50, and 75% shading riparian vegetation, and (3) quantify the relations among riparian vegetation and changes in thermal responses during periods of maximum stream temperatures.

# LITERATURE REVIEW

## Wisconsin Trout Stream Management

Wisconsin has over 16,690 km of trout streams classified into three groups based on the level of natural reproduction, trout presence, and stocking (WI DNR 2002a). Class 1 waters, of which there are 6,656 km, are defined as having natural reproduction that sustains a trout population at or near the carrying capacity of the stream and require no stocking. Class 2 waters are classified as having some natural reproduction and year to year survival but require stocking to support a preferred level of fishing opportunities and account for 7,474 km of Wisconsin's trout streams. There are 2,560 km of Class 3 trout water which have no natural reproduction or year to year survival and require annual stocking to provide trout fishing opportunities.

Wisconsin trout stream management consists of regulations, land acquisition/protection, stocking, and habitat management. Wisconsin's inland trout regulations are based in part on stream productivity, fishing pressure, and the anglers' desires (WI DNR 2002a). Inland trout fishing season consists of a regular open season for all waters, including lakes, beginning the 1st Saturday of May that runs through the end of September and an early catch-and-release season for trout starting at the beginning of March and ending late April. During the regular season there are four main regulation categories all waters fall under. Category two streams have the most liberal regulations with a minimum size (total length) of 17.8 cm and bag limit consisting of a combination of five trout of any species. Category three streams have a size limit of 22.9 cm and a bag limit of a combination of three trout of any species. Category four streams consist of brown and rainbow trout minimum length limits of 30.5 cm, a brook trout length limit of 20.3 cm, and a combined bag limit of three. Category five has special regulations that are stream- or lake-specific. Ten trout can be held in possession of which only five can be a combined total of brown and rainbow trout. The early catch-and-release season is only applicable to streams and rivers in specified areas. During the early catch and release season only artificial lures can be used and all trout caught must be released immediately due to a daily bag and possession limit of zero (WI DNR 2008).

Stocking helps to provide additional trout fishing opportunities in areas where natural fish production does not realize the potential carrying capacity of a body of water whereas areas with no natural trout reproduction can provide a trout fishery. In 2008, 589,893 catchable-size brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), and rainbow trout (*Oncorhynchus mykiss*) were stocked throughout Wisconsin. These trout were reared in six state hatcheries distributed across the state. Some of these trout are considered to be "wild" as part of Wisconsin's wild trout stocking program (Mitro 2004). This program involves the release of stocked trout raised in hatcheries that are the progeny of wild parents. Stocked trout from wild parents survive in greater numbers than their counterparts that come from generations of hatchery-reared fish. The Wisconsin Department of Natural Resources hopes that these fish will help restore self-sustaining populations of trout to hundreds of kilometers of Wisconsin's streams (Avery et al. 2001). These "wild" trout could be used to restore trout populations to areas where trout were formerly found before conditions became unsuitable as a result of habitat degradation but have since been restored.

Habitat management on Wisconsin's trout streams currently focuses on modifying the stream channel and banks in order to create more "desired" habitat conditions for trout. The majority of this habitat management focuses on creating more cover for adult fish and fails to consider a major limiting factor in trout distribution and survival: water temperature. Trout require cold water and under current trout stream management, it is suggested that streams must be managed to maintain or create more cold water (WI DNR 2008). However in practice, little actual management aimed at decreasing stream temperatures and maintaining current cold water is taking place even considering that climate change is becoming a concern in coldwater fisheries management nationally (Keleher and Rahel 1996; Rahel et al. 1996; McCullough et al. 2009). Removal of beaver dams that cause water to warm in small impoundments and habitat restoration aimed at narrowing wide channels to increase depth are the only two current management practices addressing stream temperatures (Avery 1992; McRae and Edwards 1994; Blann et al. 2002). With Wisconsin having a trout stream management goal of increasing and offering better fishing opportunities, maintaining colder stream temperatures and possibly increasing the length of stream with suitable water temperatures for trout should be a management objective worth exploring in greater depth.

In Wisconsin, numerous trout stream habitat management techniques are commonly used (White and Bryndlson 1967; Hunt 1993). Habitat improvement projects consist of implementing skyhook bank cover structures, Little Underwater Neighborhood Keepers Encompassing Rheotaxic Salmonids (L.U.N.K.E.R.S), bank cover logs, current deflectors, wedge dams, and brush bundles to mention a few of the more popular techniques. L.U.N.K.E.R.S. consist of two layers of boards held together with 8-10 inches of space between them that are anchored into the streambed along the outside bend of streams (Hunt 1993). The stream bank is then sloped over the top and graded to provide overhead cover and increase pool habitat. Brush bundles involve the removal of streamside woody vegetation, usually tag alder (*Alnus rugosa),* to be anchored to the inside of meanders (i.e., non-current bearing sides) of streams in order to trap sediment and create narrower and deeper stream channels at normal flows (Hunt 1993). There are several guidelines for determining what type of habitat structure to use under ambient stream conditions (e.g., either for low- to moderate-gradient streams or high-gradient stream). But, there is a lack of any formal analysis for determining what habitat management technique is appropriate for specific stream conditions.

Trout habitat improvement techniques have resulted in questionable success rates (Hunt 1988; Avery 2004). In a review of 64 trout stream habitat improvement projects, Avery (2004) found that 59% of the projects achieved “some form” (e.g., post-development increases of a given population variable of 25% or greater) of successful improvement to trout populations and habitat conditions. This equivocal return in investment is compounded by the fact that the majority of these management techniques are short-term, often degrading after several years and requiring some continued upkeep. In some cases, these projects may be incorrectly or overly applied in habitat restoration attempts and thus damage stream geomorphology while in others, they may not even address actual limiting conditions. Furthermore, current improvement techniques often neglect potential improvements that may be realized through long-term riparian management strategy such as reforestation of formerly forested riparian areas (Barton et al. 1985). By focusing on reforestation of formerly forested riparian areas, a more sustainable and perhaps cheaper long-range management plan, with benefits such as decreasing water temperatures, increased secondary production, bank stabilization, and large woody debris recruitment, could be achieved.

Brook Trout Thermal Requirements

Several studies have shown the importance of temperatures influence on the distribution and abundance of trout (Binns and Eiserman 1979; Bowlby and Roff 1986; Bozek and Hubert 1992; Stoneman and Jones 2000; Wehrly et al. 2003; Isaak and Hubert 2004). Issak and Hubert (2004) fitted an asymptotic curve to biomass and densities of rainbow, brown, and brook trout in western Wyoming and eastern Idaho versus midsummer stream temperatures. They found that biomass and densities of these trout species peaked at stream temperatures of 12°C with 3°C and 21°C predicted as the x-intercepts. When determining trout biomass in various areas of southern Ontario, Stoneman and Jones (2000) found that water temperature was the single most important habitat variable for predicting trout biomass as designated in their regression-tree model. By classifying streams based on summer mean temperature and temperature fluctuations in Michigan's lower peninsula, fish species distribution patterns, community composition, species richness, and standing stocks could be predicted (Wehrly et al. 2003). In this case brook trout's optimum habitat suitability was said to occur at cold (mean weekly July temperatures <19°C) and stable (mean July weekly water temperature fluctuations <5°C) stream temperature regimes.

As a coldwater species, summer stream temperature is the most important factor that limits the distribution of brook trout throughout its range (Greene 1950; MacCrimmon and Campbell 1969). Trout are stenothermal obligate poikilothermic ectotherms (i.e. cold blooded). They survive across a narrow thermal range and are strict thermal conformers. The body temperature of trout changes to approximate water temperatures and conduction is roughly 80% responsible for this exchange (Elliott 1981; Beitinger et al. 2000). Their body temperatures are usually less than 0.6°C above the water temperature. Moreover, this body temperature does tend to increase slightly with increasing size and weight (Elliott 1981). Temperature affects brook trout reproduction, disease and parasite susceptibility, predation vulnerability, feeding, growth, respiration, heartbeat, digestion, and movement, the latter of which are effected by metabolism (Fry 1947; Brett 1956; Brett 1971; Coutant 1973; USEPA 1976; Elliott 1981).

Trout reproduction is limited by temperature as it affects gonadal development, spawning, and embryo survival and development (USEPA 1976). Optimal temperatures for spawning and incubation occur at approximately 9°C and range from 5 - 13°C for brook trout (Frost and Brown 1967; Hokanson et al. 1973; USEPA 1976). Elevated temperatures (> 19°C) cause severe decreases in the sexual maturation of trout and can even halt maturation all together. In general, brook trout do not inhabit water temperatures greater than 19°C, except for brief periods of time, or mean monthly temperatures greater than 16°C for a month prior to spawning to insure proper sexual maturation (Hokanson et al. 1973). Spawning may be interrupted or aborted if temperatures are too high, and the number of spawning adults decreases progressively with increasing temperatures (Hokanson et al. 1973). Even after successful spawning, the eggs are very susceptible to increased water temperatures because warmer temperatures can prevent embryonic cleavage during development and drastically decrease the percent of normal hatch. When brook trout eggs are incubated in temperatures greater than 12°C, normal hatch declines from roughly 60 to 0% at 15°C (Hokanson et al. 1973).

Parasitism and disease increase in brook trout as temperature increases. This occurs in trout partially due to the fact that they congregate in more limited areas of thermal refuge, causing overcrowding, which leads to more disease transmission (Hokanson et al. 1977; Coutant 1987). Increased temperatures can also lead to thermal stress in fish, which can result in an increased incidence of disease or parasitism and decrease their ability to withstand diseases resulting in increased mortality (Wedemeyer and McLeay 1981; Materna 2001; Williamson 2007). Fryer and Pilcher (1974) found that rainbow trout, coho salmon (*Oncorhynchus kisutch*), and chinook salmon (*Oncorhynchus tshawytscha*) infected with *Chondrococcus columaris* and coho and chinook salmon infected with *Ameromonas salmonicida* all had low mortality rates at 3.9 - 9.4°C, moderate rates at 12.2°C, and high mortality rates at 17.8°C. Progress of the infections increased with increasing temperatures as well.

Increased water temperature increases food and oxygen consumption needs in order for trout to physiologically sustain themselves (Fry 1948; Graham 1949; Frost and Brown 1967; Elliott 1976; Hughes et al. 1978). Higher temperatures create greater metabolic costs and thus more energy is needed through the consumption of food and oxygen uptake. These food requirements continue to increase with temperature until the upper incipient lethal temperature is reached; the trout eventually die unless cooler, more suitable temperatures are attained (Hughes et al. 1978). Food needs may continue to increase with increasing temperatures but feeding rates may not after a certain temperature is reached. For instance, brook trout food consumption increases two-fold for every 4°C increase in water temperature until 13°C; at 17°C feeding begins to decrease and at 21°C they only consumed 6% of their body weight per week (Baldwin 1957). Trout have been shown to drastically decrease or cease feeding when temperatures become too high but are still within their thermal tolerance zone (Elliott 1981). This can be correlated to the increasing metabolic costs with increasing temperatures and the decreasing scope of activity for brook trout for temperatures greater than 19°C (Graham 1949; Brett 1956). Due to the fact that few natural populations feed at maximum rates, temperature limits should be shifted toward cooler temperatures than those found in many lab studies (Brett et al. 1969; Brett 1971). The oxygen uptake of an individual trout increases with increasing temperature when the fish is at rest, but when the fish is active the oxygen uptake is maximized when the metabolic peak is reached. This occurs around 18 - 20°C for brook trout which is before their upper thermal limit (Graham 1949). The problem with this trend is that as oxygen requirements for trout increase with temperature, the overall concentration of oxygen in the water decreases with temperature, acting as a limiting factor as a result of increased temperatures. Overall, increased temperatures can make it harder for a brook trout to sustain themselves and grow due to food limitations in natural settings and the need for more oxygen in warmer oxygen deficient waters.

Behavioral thermoregulation is the only real means for trout to deal with unsuitable water temperatures. This involves the use of thermal refuges, such as colder tributary streams, deeper pools, and areas where large groundwater inputs are available (Kaya et al. 1977; Bowlby and Roff 1986; Berman and Quinn 1991; Matthews and Berg 1997; Torgersen et al. 1999; Baird and Krueger 2003). Brook trout in a fifth-order Adirondack river in New York had body temperatures 4.0°C lower than the main flow of the river when it reached temperatures greater than 20°C. These brook trout were found to be using cooler tributary confluences or groundwater discharge areas within pools in the main river (Baird and Krueger 2003). Otherwise, they must migrate to stream reaches with more preferable temperatures or perish (Neil and Magnuson 1974; Beitinger and Fitzpatrick 1979; Neil 1979; Reynolds and Casterlin 1979; Meisner 1990; Kaeding 1996; Hayes et al. 1998). Ford River, Michigan brook trout migrated upstream to cooler headwater tributaries, <20.8°C, when the river maximum temperatures ranged from 22 - 23.8°C (Hayes et al. 1998).

With increased stream temperatures also comes increased competition from other fish species, such as some coolwater species, which may out-compete trout at warmer temperatures along their thermal tolerances (Reeves et al. 1987; Taniguchi et al. 1998; Reese and Harvey 2002). For instance, when water temperatures reach 22°C, creek chub (*Semolitus atromaculatus*) have been shown to out-compete brook trout; at 24°C creek chub become completely dominant (Taniguchi et al. 1998). Brown trout are found in many brook trout streams and have been shown to out-compete brook trout in the lower, warmer reaches of streams (Fausch and White 1981; Waters 1981). Brown trout have a higher metabolic peak and greater total scope of activity at higher temperatures compared to brook trout (Brett 1956). For brook trout, stream temperatures increasing above 19°C make it progressively harder to compete for food and other resources with other fishes (Brett 1956).

Studies on brook trout have identified optimum and preferred temperatures, the upper incipient lethal temperature, the critical thermal maxima, and field observation-based temperatures (Table 1). Optimum temperatures are temperatures at which the maximum amount of energy is allocated toward growth and reproduction or an activity is best performed, whereas preferred temperature is the temperature or range of temperatures that brook trout will move toward if presented with the choice (Fry 1947). In fact, it has been found that fish tend to avoid water temperatures just 1 - 3°C above their preferendum (Cherry et al. 1975). The upper incipient lethal temperature (UILT) is the temperature at which 50% of the population will no longer live for an indefinite period of time and the edge of the zone of tolerance (Fry 1947). The critical thermal maxima (CTM) is the water temperature at which fish lose their equilibrium when exposed to water that is heated or cooled at a rate that does not allow for thermal acclimation but allows the body core temperature to match the water temperature (Selong et al. 2001). Field, study-based temperatures are temperatures at which brook trout have been found to occupy in nature based on field data. Maximum weekly average temperatures (MWAT) and maximum daily average temperatures (MDAT), MWAT discriminate between trout and non-trout waters best (Barton et al. 1985; Eaton et al. 1995). Preferred temperatures during MWAT are considered to be those ≤ 19°C for brook trout (Creaser 1930; Hokanson et al. 1973; Cherry et al. 1975; Brungs and Jones 1977; Peterson et al. 1979) while suitable stream temperatures during the MWAT period are considered to be less than 22.3°C (Eaton et al. 1995), 23.3°C (Wehrly et al. 2007), and 24°C (Meisner 1990). MDAT ≤ 22°C is considered the cut off for coldwater

**Table 1.** Brook trout optimum temperature, preferred temperature, critical thermal maxima (CTM), upper incipient lethal temperature (UILT), and maximum temperature presence through field observations.

| Brook Trout Thermal Criteria | | | | |
|---|---|---|---|---|
| | Temperature($^{o}$C) | Acclimation temperature ($^{o}$C) | Life Stage | Reference |
| Optimum | | | | |
| | 13 | | Yearling | Baldwin (1957) |
| | 13 - 16 | | | Dwyer et al. (1983) |
| | 15.4 | | | Brungs and Jones (1977) |
| | 16 | | Juvenile | Graham (1949) |
| | 15.4 | | Alevins Swimming | McCormick et al. (1972) |
| | 16.1 | | Adult | Hokanson et al. (1973) |
| Preferred | | | | |
| | 18.2 (max 7-d mean) | | | Brungs and Jones (1977) |
| | 16.5 | Final Preferendum | Juvenile | Cherry et al. (1975) |
| | 11.2 | 6 | Juvenile | Cherry et al. (1975) |
| | 11.3 | 9 | Juvenile | Cherry et al. (1975) |
| | 13.7 | 12 | Juvenile | Cherry et al. (1975) |
| | 15.2 | 15 | Juvenile | Cherry et al. (1975) |
| | 18 | 18 | Juvenile | Cherry et al. (1975) |
| | 18.3 | 21 | Juvenile | Cherry et al. (1975) |
| | 19 | 24 | Juvenile | Cherry et al. (1975) |
| | 16 | Final Preferendum | Juvenile | Cherry et al. (1977) |
| | 19 (max) | | | Creaser (1930) |
| | 14 - 19 | | Juvenile | Graham (1949) |
| | 10 - 19 | | Adult | Hokanson et al. (1973) |
| | 17.5 | 12.1 | Fingerlings | Peterson (1979) |
| | 8.7 | 12.7 | Fry | Peterson (1979) |
| Upper incipient lethal temperature (Selong et al. 2001) | 24 (7-d) | 24 (1$^{o}$C /day) | Juvenile | Cherry et al. (1977) |
| | 23.5 | 3 | Juvenile | Fry et al. (1946) |
| | 24.6 | 11 | Juvenile | Fry et al. (1946) |
| | 25 | 15 | Juvenile | Fry et al. (1946) |
| | 25.3 | 20 | Juvenile | Fry et al. (1946) |
| | 25.5 | 22 | Juvenile | Fry et al. (1946) |
| | 25.5 | 24 | Juvenile | Fry et al. (1946) |
| | 25.5 | 25 | Juvenile | Fry et al. (1946) |
| | 20.2 (7-d) | 12 | Alevins Hatched | McCormick et al. (1972) |
| | 24.5 (7-d) | 12 | Alevins Swimming | McCormick et al. (1972) |
| Critical thermal maxima (Selong et al. 2001) | 29 | 10 | Juvenile | DeStaso and Rahel (1994) |
| | 28.7 - 29.8 | 10-20 | Juvenile | Lee and Rinne (1980) |
| | 28.3 - 30.8 | 8-20 | | Selong et al. (2001) |
| Field, study-based | 25.6 (max) | | | Barton et al. (1985) |
| | 24.2-26.3 (max) | | | Binns and Eiserman (1979) |
| | 26.5 (max) | | | Bowlby and Roff (1986) |
| | 22.3 (max 7-d mean) | | | Eaton et al. (1995) |
| | 24 (max 7-d) | | | Meisner (1990) |
| | 24 (max) | | | Picard et al. (2003) |
| | 23.3 (max 7-d mean) | | | Wehrly et al. (2007) |

streams (i.e, brook, brown, and rainbow trout) (Lyons et al. 1996).

Stream Thermal Regime

Water temperature is proportional to the heat load, a quantity of heat energy added to the water, divided by the stream discharge. With many factors influencing the quantity of heat energy in a stream and discharge of a stream, the thermal regime of streams can be quite complex and stream energy budgets very dynamic. Theurer et al. (1984) and Moore et al. (2005) provide a more comprehensive review on each individual factor influencing stream temperatures. Thermally influential factors include solar radiation, long-wave radiation (e.g., atmospheric, vegetative, and radiation emitted by the water), convection, evaporation, and streambed conduction (Figure 1). These underlining physical properties are most markedly altered by discharge, shade, air temperature, windspeed, stream morphology, groundwater inputs, and riparian conditions (Theurer et al. 1984; Sullivan and Adams 1991; Walks et al. 2000; Poole and Berman 2001; Gaffield et al. 2005). Hyporheic flow, the flow and dynamics of flow in the zone beneath the stream bed where shallow groundwater and surface water mix, is another factor influencing stream temperatures (Poole and Berman 2001; Burkholder et al. 2008).

As thermal inputs are altered, the stream equilibrium temperature (i.e., the stream temperature achieved when the interaction of all heat fluxes through the water surface counterpoise each other to sum zero) will change and the stream temperature will begin to approach a new equilibrium temperature. Some streams may never achieve an actual equilibrium temperature due to how often the stream temperature driving factors change resulting in rapid changes in equilibrium temperatures. An example of this change in equilibrium temperature exists during summer days where the equilibrium temperature of open

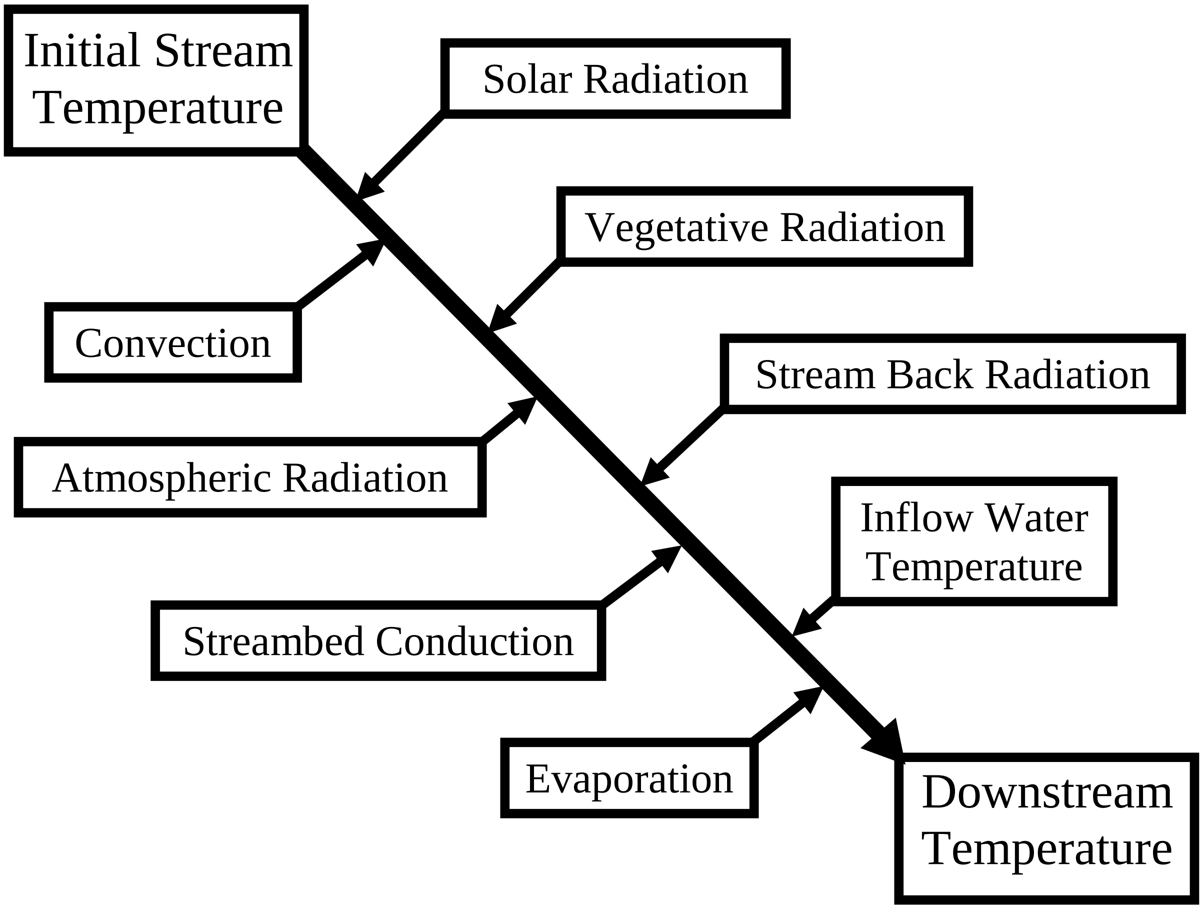



**Figure 1.** Factors contributing to stream temperature change from initial upstream temperature to final downstream temperature.

stream reaches is higher than that of shaded reaches (Figure 2) (Theurer et al. 1984; Gaffield et al. 2005).

In central Wisconsin, brook trout streams are highly groundwater fed. Groundwater inputs are the only way these streams can attain suitable temperatures for trout because they lack year-round snowmelt and occur at low elevations and low latitudes which are warmer. Stream headwaters start where groundwater is discharged and thus are colder and warm as distance downstream increases (Figure 2). In areas of high groundwater input, this warming is not as great of concern to trout because the warming is mitigated by the cold groundwater inputs. However, as the ratio of groundwater inputs relative to total discharge decreases downstream, other factors influencing stream temperature start to play a greater role. Of the factors influencing stream temperature, solar radiation plays one of the most significant roles in stream warming during summer (Brown 1969; Johnson 2004; Moore et al. 2005). It is this factor that is the easiest to manage by increasing the amount of shade provided to a stream through riparian vegetation management.

## <u>Land Use and Streams</u>

### *Historical*

Brook trout likely encompassed a much wider distribution in pre-settlement times (Greene 1935; Becker 1983). Excluding wetlands, 95% of Wisconsin was once almost entirely covered by virgin forest (Verry and Dolloff 2000). Only uplands in southern portions of the state contained grasslands (Curtis 1959; Verry and Dolloff 2000). In these areas, forests were confined to streambanks and lake shores (Curtis 1959). Since the time of early European settlement, most of the landscape has changed due to logging, agricultural practices, and

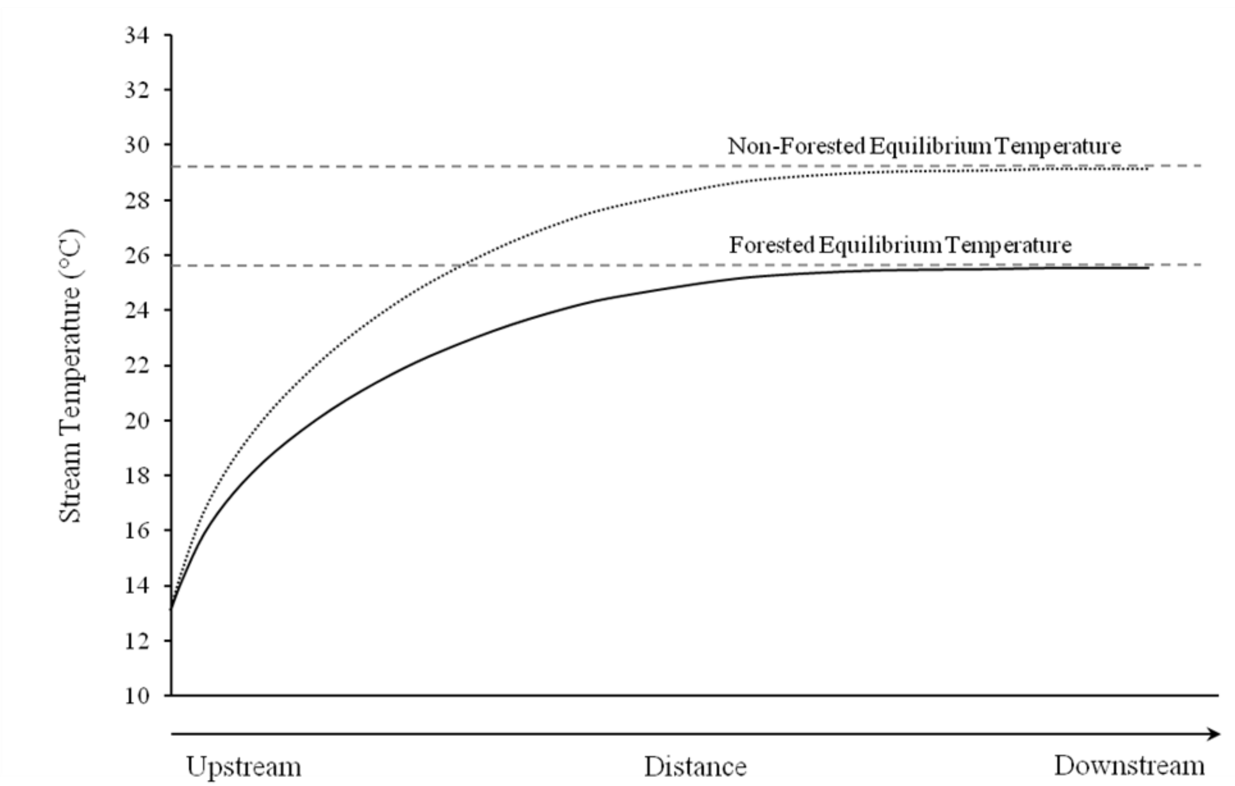



**Figure 2.** Typical longitudinal stream temperature profile showing equilibrium temperature differences between forested and non-forested stream reaches.

urbanization (Curtis 1959; Knox 1977; Verry and Dolloff 2000; Wang et al. 2003a). Because strong links exist between land use, stream fish assemblages, and fish habitat quality, much of the former brook trout streams in Wisconsin were lost due to these land use conversions (Wang et al. 1997). Lyons (J. Lyons, Wisconsin Department of Natural Resources, personal communication) estimated that 45% of brook trout streams were lost in Wisconsin from 1850 to present. Greene (1935) stated that brook trout were rarely abundant in any Wisconsin stream due to the land degradation by early European settlers and overfishing. Becker (1983) also noted that central and southern Wisconsin's populations of brook trout seemed to be decreasing in range and became extirpated in some areas.

One reason for declines may have been the increase in stream temperatures as a result of modified land use, specifically, with the conversion of old-growth forest dominated landscapes to agriculture that occurred over this time period (Curtis 1959; Verry and Dolloff 2000). These land use changes decreased shading and altered microclimates that kept streams cooler and changed groundwater recharge dynamics in upland watershed areas. Increases in stream temperature have been documented in many different regions of the state as the result of land use practices (Wang et al. 2003a, Wang et al. 2003b). The major loss of thermally suitable streams for trout in Wisconsin likely occurred in the North Central Hardwoods Forests Ecoregion in central Wisconsin and the Southeastern Wisconsin Till Plains Ecoregion because these areas were and remain the areas modified to the greatest magnitude (J. Lyons, Wisconsin Department of Natural Resources, personal communication).

*Stream Degradation*

Since European settlement, virtually the entire state of Wisconsin has been clearcut logged including riparian areas (Curtis 1959; Verry and Dolloff 2000; Wisconsin Woodland

Owner's Association 2004). Riparian clearcut logging is no longer allowed because it negatively affects streams in many ways (Chamberlin et al. 1991; WI DNR 1995) such as changing channel morphology and stream habitat, decreasing organic matter inputs, and increasing peak flows, sedimentation, and stream temperatures (Chamberlin et al. 1991; Hornbeck and Kochenderfer 2000). These changes in turn adversely influence stream conditions and stream biota.

After lands in Wisconsin were cleared of timber, many families converted the land to agriculture (Carstensen 1958). Currently, 43% of Wisconsin land is managed with agricultural practices such as row cropping and grazing which alters the riparian area and can lead to degrading streams conditions (Platts 1981; Kauffman and Krueger 1984; Platts 1991; Wohl and Carline 1996; Wang et al. 1997). Land use conversion from forests to agriculture can increase the flood magnitude and duration, stream sediment inputs, reduced base flows, as well as create wider and shallower stream channels for smaller streams when compared to historic conditions (Knox 1977; Chamberlin et al. 1991). Wider and shallower streams can increase stream temperature by increasing the amount of solar radiation received by a stream and increasing the amount of convection taking place with warmer summer air temperatures.

Stream altering factors, such as livestock grazing, can take years to recover from and drastically alter fish habitat, but they can recover over time (Platts 1991). Livestock grazing in riparian areas has reduced suitable fish habitat and trout populations in many streams throughout North America (Keller et al. 1978; Platts 1981; Platts 1991). Grazing affects streamside vegetation, increases sedimentation, changes channel morphology (e.g., substrates, pool to riffle ratios, bank sloughing, altered planform, and wider and shallower channels), and decreases water quality (e.g., increased water temperatures, increased suspended sediments, and increased nutrient levels) (Platts 1991). Studies have shown large increases in trout populations when grazing has ceased and larger trout populations occur in non-grazed sections of streams compared to grazed sections (Lorz 1974; Armour 1977). Keller and Burnham (1982) found increased trout habitat in areas of Summit Creek, Idaho where cattle were fenced out. Excluding cattle led to decreased stream width, increased depth and pool quality, increased bank cover, and increased bank stabilization. Several years after the cattle exclusions were implemented, non-grazed sections of the stream proved to hold more and larger trout than grazed sections (Platts 1991).

Urban land use can also affect stream fish assemblages. Urbanization reduces suitable trout stream habitat and increases tolerant fish species (Wang et al. 1997; Stewart et al. 2001; Wang et al. 2003a). Water temperatures increase as percent urbanization land use increases and base flow decreases (Wang et al. 2003a). As a result, trout habitat suitability declines because trout require colder water temperatures and greater base flow.

Several studies have assessed the potential impacts climate change might have on stream temperatures (Meisner 1990; Sinokrot et al. 1995; Eaton and Sheller 1996; Poff et al. 1996; Pilgrim et al. 1998; Schindler 2001). As greenhouse gasses in the atmosphere increase resulting in increased air temperatures and greater variability in precipitation, there could be major implications to coldwater fisheries (Sinokrot and Stefan 1993). Increases in air temperature could cause increased groundwater temperatures leading to warmer initial stream temperatures (Meisner 1990) and directly increase stream temperature warming (Pilgrim et al. 1998). Climate change is also expected to alter streamflows which can adversely affect stream temperatures (Poff et al. 1996). Eaton and Scheller (1996) predicted a 54.8% reduction in brook trout range throughout the United States based on projected mean summer air temperature increases of 4.4°C throughout the U.S. In the model created by Keleher and Rahel (1996), Wyoming's salmonid range is predicted to decrease 16.2% if the mean July air temperature increases 1°C and 68.0% if it increases by 5°C. Population fragmentation would also occur as trout are forced into headwater refugia in stream drainages which provide suitable water temperatures (Rahel et al. 1996).

## Riparian Vegetation Function

Riparian vegetation helps structure stream morphology and fish habitat and plays a key role in stream productivity and suitable water chemistry for certain biotic communities (Gregory 1992; Wang et al. 1997; Lyons et al. 2000). For fish managers, riparian vegetation management is one of the most basic and applied management techniques they can use. Examples of riparian vegetation management to benefit fisheries have and, still are, taking place in Wisconsin. Managing for riparian grasses to narrow and deepen streams in the southwest (Trimble 1997, Stephens 2001), removing aspen from streams in the north to prevent beaver from inhabiting streams (McRae and Edwards 1994), and removing woody riparian vegetation to increase trout biomass (Hunt 1979) are a few examples. Applications of specific riparian vegetation management techniques should be site-specific taking into account the most limiting factors of the species being managed.

### *Current Riparian Vegetation Management*

In Wisconsin, best management practices (BMPs) for current riparian vegetation management includes harvesting woody vegetation at a minimum of ten year intervals and leaving at least 13.77 square meters of basal area per hectare in trees 12.7 cm diameter breast height and larger evenly distributed (WI DNR 1995). These riparian area management zones are 30.5 m landward for lakes and navigable perennial streams and 10.7 m landward for intermittent streams. However, their use is for current forestry practices and not other land use activities, and private land owners are not required to follow these BMPs. Restoration and/or maintenance of riparian areas in agriculture and urbanized areas are encouraged in the BMPs. It is also suggested that riparian areas being grazed are done with great precaution as heavy grazing in the riparian area can have many negative impacts on stream quality and integrity (Platts 1991).

Management of trout stream riparian areas varies. One popular riparian vegetation management technique in Wisconsin trout streams calls for the removal of shrubs and trees in opt of managing grasses (Hunt 1993). Grasses in these areas are then maintained by limited livestock grazing, prescribed burns, or continued manual removal (Hunt 1993; Lyons et al. 2000; Stephens 2001). It is believed that grasses increase the amount of undercut banks, decrease stream width resulting in increased stream depths and velocities, decrease erosion, and increase primary production in the stream resulting in more food opportunities for trout (Hunt 1979; Lyons et al. 2000). This technique originated on the Little Plover River, Lunch Creek, and Spring Creek located in central Wisconsin (Hunt 1979). It is perceived to be a successful management technique as long as elevated stream temperatures from increases in solar radiation do not limit the stream’s thermal suitability for trout (Hunt 1979).

Riparian vegetation is also managed through government programs such as the Environmental Quality Incentives Program (EQIP), the Conservation Reserve Program (CRP), the Riparian Forest Buffer Program, and the Wisconsin Conservation Reserve Enhancement Program (CREP). Currently, the Wisconsin Department of Natural Resources (WI DNR) has trout stream watershed management high on its priority list. This watershed management consists of protecting against watershed degradation by having the Department regulate its permitting system around trout streams and through education and cooperation with the public by using the priority watershed program (WI DNR 2008). WI DNR also purchases and leases land along trout streams for protection against harmful land management and to provide fishing opportunities (WI DNR 2002a).

*Primary and Secondary Production*

Vegetation in the riparian area can influence primary and secondary production occurring in stream and thereby influence trout production. Riparian trees provide food for some aquatic organisms because of increased allochthonous material inputs from organic debris while open grass reaches benefit streams by producing more autochonous materials (Lyons et al. 2000). Allowing light energy to reach the stream in open areas leads to higher microbial respiration associated with organic matter, increased primary production, and greater production of invertebrates providing more food for trout (Hunt 1979; Murphy and Hall 1981; Murphy et al. 1981; Schlosser and Karr 1981). However, other studies have suggested that there is greater abundance of micro-organisms and macroinvertebrates in forested areas. This is likely due to the organic inputs provided by trees and large woody structure (i.e., debris). These inputs can amplify their effect by trapping organic material allowing microorganisms and detritivores the chance to decompose them (Gregory et al 1991; Sweeney 1993). Woody structure in streams with sandy substrate plays a more crucial role in production by providing habitat and food for the invertebrates (Wallace and Benke 1984; Benke et al. 1985).

*Bank Stabilization, Sediment Reduction, and Channel Morphology*

The ability of riparian vegetation to stabilize banks has been widely studied (Trimble and Saartz 1951; Thorne 1990; Simon and Collinson 2002; Wynn et al. 2004). Simon and Collinson (2002) found increases in soil cohesion ranging from 2 to 18 kPa in the upper meter of soil from various vegetation roots increasing streambank resistance to erosion. With different types of vegetation offering different bank stabilization and sediment reduction properties, managers must closely review what type is best for each stream based on its individual characteristics. There are limitations to when and where certain types of vegetation will and can grow adding more aspects that managers must take into account.

Grasses, shrubs, and trees all have different root properties which are species dependent and therefore have different stabilization properties (Simon and Collinson 2002; Wynn et al. 2004). Since soil is strong in compression and weak in tension and roots are strong in tension and weak in compression, root-laden soil enhances bank stabilizing properties by increasing the overall soil strength. Vegetation also decreases the pore-water pressure by extracting soil moisture through transpiration leading to soil strengthening by matric suction (Simon and Collinson 2002). Generally grasses have the highest root densities near the surface offering more stabilization in the top portions of the soil; trees have larger and greater root densities at deeper depths offering greater stability at greater depths. When stabilizing the banks of small streams, grasses might provide better results, but as streams widen, deepen, or have steeper banks trees tend to offer greater bank stability (Thorne 1990; Lyons et al. 2000; Simon and Collinson 2002; Wynn et al. 2004).

Stream width is influenced by riparian vegetation. Small streams with smaller watersheds tend to be narrower in reaches with riparian grasses compared to forested reaches due to grass's ability to protect banks against fluvial scour and trap alluvium (Davies-Colley 1997; Anderson et al. 2004). When stream watershed size increases, the narrowing effect of grasses decreases and streams will eventually become narrower in forested reaches (Davies-Colley 1997; Anderson et al. 2004). Streams with watersheds less than 1 $km^2$ are twice as wide in forested verses grass reaches (Davies-Colley 1997). Stream channels with watersheds reaching 10 to 100 $km^2$ are actually narrower with thick woody, stream-side vegetation compared to grass and non-forested riparian areas (Anderson et al. 2004). In Coon Creek located in southeast Wisconsin, Trimble (1997) found streams to be narrower in grass reaches of streams compared to forested reaches. In one study, central Wisconsin stream widths were not found to be significantly different between wooded and grassed riparian areas (Stephens 2001).

Substrate and sediment in streams can be highly influenced by the streamside vegetation. Sediment received by streams can be reduced by riparian vegetation because of its ability to trap sediment (Trimble and Sartz 1957). The amount and efficiency of sediment trapping before it reaches a stream can depend on vegetation type and site specific variables, but all vegetation types do very well at filtering sediment and nutrients (Schlosser and Karr 1981; Lowrance et al. 1984; Cooper et al. 1987; Parsons et al. 1994; Daniels and Gillman 1996; Wenger 1999). In North Carolina, buffer strips on drainage ditches in agriculture fields reduced the total amount of incoming sediment by 80% (Daniels and Gillman 1996). The distribution and type of substrate can also vary with riparian vegetation type. Willow branches can trap silt and debris and increase sediment deposition (Hunt 1993). Trees can contribute woody debris that can cause scouring and expose gravel and other larger substrates as well as decrease velocities in other areas leading to smaller sediment being locally deposited creating more substrate diversity (Gurnell et al. 2002).

Riparian trees and shrubs are removed throughout much of Wisconsin as a trout stream management technique. Stream banks are managed for grasses in hopes of creating more stable banks, narrower stream widths, and increasing primary production (Hunt 1979). This may occur to some degree but studies need to be conducted in order to determine when grasses might provide these desired effects. Stream channel erosion and width-to-depth ratio in southwestern Wisconsin is greater in areas of riparian forests compared to grasses (Trimble 1997; Stephens 2001). However, in central Wisconsin no significant difference in stream width was found between grass and wooded riparian vegetation type, suggesting that riparian vegetation should not be managed with stream width as a concern in this geographic area (Stephens 2001).

*Stream Habitat*

Stream habitat can be influenced by riparian vegetation (Angermeier and Karr 1984; Wesche et al. 1987; Beschta 1991; Hicks et al. 1991; Swanston 1991), as different types of vegetation provide different types of stream habitat. Grasses possess dense and shallow root systems that can create undercut banks (Hunt 1979; Davies-Colley 1997). These undercut banks provide hiding, resting, and prey-stalking cover for brook trout (Hunt 1979; Riley et al. 1992; Peterson 1993) and other species of fishes (Rabeni and Sowa 1996). Sections of stream with little shading have greater primary production as a result of increased light levels and produce greater macrophyte growth which fish can use as cover (Hunt 1979; Hunt 1993; Peterson 1993; Thorn and Anderson 1993). Large woody debris and rootwads recruited to streams from riparian trees provide a substantial source of habitat for fish and other aquatic organisms (Bryant 1983; Angermeier and Karr 1984; Sedell et al. 1988; Hunter 1991; Sundbaum and Näslund 1998; Wang et al. 1998).

*Stream Temperature*

Much research has been focused on quantifying and characterizing riparian vegetation impacts on stream temperatures (White 1996; Poole and Berman 2001; Moore et al. 2005; Webb et al. 2008). Different types and amounts of riparian vegetation can create various microclimates, influence the amount of solar radiation a stream receives, and alter groundwater temperature inputs, which influences the thermal regime of a stream. Forest microclimates are cooler than non-forested areas in the daytime and slightly higher at nighttime leading to lower air temperature, decreased air temperature fluctuations, and cooler water temperatures (Moore et al. 2005). Riparian trees also offer greater shelter from the wind compared to grasses with there being only a fraction of the wind speed in forests compared to open areas (Stefan and Sinokrot 1993; Castelle et al. 1994; Brosofke et al. 1997; Moore et al. 2005). Lower maximum air temperatures and lower wind speeds result in lower water temperatures because the heat energy received by convection is not as great. Relative humidity in forests can be up to 25% greater versus open areas (Moore et al. 2005), and this higher relative humidity combined with lower wind speeds in forests means less heat energy lost by evaporation in a stream. The riparian microclimate is influenced by the stream as well with streams acting as heat sinks in the day, meaning the microclimate is more moist and cooler next to streams (Moore et al. 2005). Microclimates in forested riparian buffers can be greatly influenced by the buffer width. Wider buffers result in greater changes in microclimate conditions up to about one tree height for most factors (Moore et al. 2005).

Solar radiation received by a stream is one of the most influential factors affecting stream temperatures. Riparian vegetation canopies promote shading, thereby reducing solar radiation received by a stream leading to lower summer water temperatures and a reduction in stream temperature fluctuations (Brown 1969; Moring and Lantz 1975; Barton et al. 1985; Brown 1985; Sullivan and Adams 1991; Hetrick et al. 1998; Johnson 2004). Johnson (2004) reported maximum stream temperatures increased by 3-4°C just 200 m downstream of a forested riparian area when no shade was provided to a 2.1-3 m wide section of bedrock stream in Oregon. In contrast, when fully shaded, the temperature was up to 1°C lower 200 m downstream. Grasses can reduce the rate at which stream temperatures increase if they help create narrower stream channels and reduce solar radiation (Blann et al. 2002; Gaffield et al. 2005). Solar radiation heat energy absorbed by the soil can influence stream temperatures as well. The less solar radiation received by soil results in lower soil temperatures (Chen et al. 1995; Brosofske et al. 1997). Lower soil temperatures can result in lower shallow-groundwater temperatures flowing in and mixing with the stream resulting in greater stream cooling (Brosofske et al. 1997). Cooler shaded soils combined with the cooler shaded substrate in the stream leads to less heating of streams through conduction.

Buffer Widths

In Wisconsin, recent studies have suggested that riparian buffers are the most important factors for mitigating negative impacts urban and agriculture land uses in the watershed have on streams (Stewart et al. 2000; Wang et al. 2003a). The importance of riparian vegetation buffers is well established (Trimble and Sartz 1957; Barton et al. 1985; Castelle et al. 1994; Todd 2000;), but there is little consensus on the width buffers need to be to effectively mitigate other upland land use practices in the watershed. However, it is well agreed upon that larger buffer widths and lengths lead to greater stream quality and have significant positive effects on the stream biotic community (Stewart et al. 2001). Minimum buffer width requirements to protect a stream against sediment and nutrient inputs, elevated water temperatures, bank erosion, and excessive flooding depend upon many factors including soil type, slope, vegetation type, climate, and the level of upland disturbance (Trimble and Sartz 1957; Barton et al. 1985; Castelle et al. 1994; Brosofke et al. 1997; Todd 2000; Wilkerson et al. 2006). When maintaining colder water temperatures, buffers provide three main benefits. First, they tend to maintain a cooler microclimate compared to upland temperatures. In order to "maintain a pristine microclimate" around a stream, a buffer width of 45 m is needed (Brosofke et al. 1997). The second effect of buffer width on stream temperature is from shading. Mature tree vegetation buffer widths of 5 to 10 m provide the same amount of shade to a stream as wider buffers and are considered to be sufficient to attain maximum shading benefits (Todd 2000). Lastly, shelter from wind provided by riparian vegetation can alter stream temperatures (Stefan and Sinokrot 1993). However, no specific buffer width recommended to assess the impacts of wind sheltering has been reported. Wilkerson et al. (2006) looked specifically at assessing buffer width and stream temperature and found that buffer widths of 11 m showed minor but not significant increases in water temperature compared to completely forested streams; buffer widths of 23 m showed no difference in water temperature. Lengths of buffers have been shown to have affects on stream temperature as well, with longer buffer strips leading to lower temperatures (Barton et al. 1985). Barton et al. (1985) determined that the portion of riparian area that was forested 2.5 km upstream of a site elucidated 56% of the variation in weekly maximum temperature.

Stream Temperature Models

Stream temperatures are the major factor controlling the longitudinal distribution of trout and their ability to survive. Land uses and specifically riparian land uses, greatly affect those stream temperatures. Many stream temperature models have been created to assess these impacts of land use on stream temperature. However, accurate modeling of stream temperatures can be very difficult to accomplish due to the sheer complexity of all the factors influencing stream temperatures, the complexity of different stream systems, and the challenge of measuring these factors accurately.

Two main types of stream temperature models have been used to predict stream temperatures, empirical and physics-based heat budget models. Empirical models utilize measured observations to create regression or harmonic analyses to predict stream temperatures. These models are not as common as heat budget models but can predict stream temperature fairly well in the areas for which they were developed (Sullivan et al. 1990; Mitchell 1999; Mellina et al. 2002). The major advantages of empirical models are their simplicity. These models often have much simpler mathematical equations and generally fewer data input requirements as they are spatially less complex. As a result, however, these models can possess a great deal of uncertainty when used in areas they were not created for and in conditions they were not created under (i.e., different ecoregion).

Physics-based heat budget models have been developed to predict stream temperatures. Some of these models require more inputs than others but all of them are based on heat energy budget balance. These models are all based on the equation:

$$\Delta S = Q_{NR} \pm Q_E \pm Q_C \pm Q_H \pm Q_A$$

Where: $\Delta S$ is the net change in energy stored

$Q_{NR}$ is the net thermal radiation flux

$Q_E$ is the evaporative flux

$Q_C$ is the conductive flux

$Q_H$ is the convective flux.

$Q_A$ is the advective flux

Brown (1969) was one of the first to use energy budget techniques to predict temperatures of streams. He predicted hourly changes in stream temperature of 0 to 8.89°C as accurately as ±0.56°C 90% of the time in three small western Oregon streams (Brown 1969). Since Brown

(1969), many more physics-based heat budget models have been developed including: the stream network temperature model (SNTEMP) (Theurer et al. 1984), stream segment temperature model (SSTEMP) (Bartholow 1989), the enhanced stream water quality model (QUAL2E) (USEPA 1995), the Tennessee Valley Authority (TVA) river modeling system (RQUAL) (Hauser and Walters 1995), the Washington Department of Natural Resources Timber/Fish/Wildlife stream temperature model (TFWTEMP) (Doughty et al. 1991), and the TEMP-84 model (Beschta 1984). Stream temperature models previously used in Wisconsin are SNTEMP and SSTEMP by Blann et al. (2002) and a modified spreadsheet format version of SNTEMP created by Gaffield et al. (2005). Gaffield et al. (2005) reported modeling four streams in the Driftless Ecoregion of southwest Wisconsin typically within 1°C accuracy and Blann et al. (2002) reported predicting stream temperatures in Wells Creek, a tributary of the Mississippi River in the Driftless Ecoregion of southeast Minnesota, with a maximum error of 1.3°C from recorded stream temperatures.

<u>Summary</u>

Brook trout range throughout Wisconsin has been reduced compared to historical distributions. Much of the reduction in brook trout inhabited streams is believed to be due to land use changes that have led to increased summer stream temperatures. Elevated water temperatures have reduced suitable brook trout habitat throughout Wisconsin. Brook trout tend to prefer temperatures ranging from 10 to 19°C and suitable temperatures are considered to be those less than 22.3°C for the MWAT period. If corrected, riparian vegetation offers potential for mitigating the increased temperatures caused by poor riparian land use thereby providing an opportunity to increase the amount of stream with suitable temperatures for brook trout. Trees in the riparian area offer great potential to reduce stream temperatures to suitable levels for trout and increase the length of stream suitable to brook trout by adding shade to the stream and altering the microclimate to promote cooler stream temperatures. Riparian forests also offer a variety of other benefits to streams including bank stabilization, creating in-stream trout habitat, increasing overall water quality, and providing organic inputs into the stream for increased secondary production. Adding forested buffer strips to streams can help restore coldwater streams by increasing stream shading resulting in a decreased amount of shortwave radiation and influence the riparian microclimate. To fully restore temperatures of coldwater streams major land use changes have to occur at a watershed level. However, it has been shown that riparian land use is most strongly linked to stream temperature changes and therefore much of the coldwater stream temperatures can be restored by correctly managing these areas.

## METHODS

### Study Sites

Twelve study streams were chosen for this study: Unnamed 17-5 / Cunningham Creek in Clark County, Blake Creek and Little Wolf River (North Branch) in Waupaca County, Magdanz / Hatton Creek and Sucker Creek in Waushara County, and West Branch Shioc River in Shawano County were assessed in the summer of 2007. In the summer of 2008, Bronken Creek and Spring / South Branch Pine Creek in Dunn County, Gillespie Creek in Polk County, Onemile Creek and Webster Creek in Juneau County, and Walla Walla Creek in Waupaca County were surveyed (Table 2; Figure 3). Stream selection was based on three main criteria:

1. Streams transitioned from coldwater Class 1 or 2 trout water (naturally reproducing brook trout streams) systems to non-classified or class 3 waters (no natural reproduction of brook trout) before the confluence with a large river.
2. Streams were less than six meters in width.
3. Streams were located in the North Central Hardwoods Forest Ecoregion of Wisconsin.

### Data Collection

All streams were surveyed to obtain the most appropriate data for use in stream temperature prediction models while using common survey techniques (Platts et al. 1987; Rosgen 1994; Simonson et al. 1994; WI DNR 2002b; Bartholow 2004; WI DNR 2004). All measurements taken for modeling occurred at segment, transect, transect point, and site specific scales.

**Table 2.** 2007 and 2008 study stream information.

| Stream Name | ID # | Length Surveyed (km) | Watershed Area ($km^2$) | County | Latitude / Longitude | Local weather station(s) |
|---|---|---|---|---|---|---|
| 2007 | | | | | | |
| Unnamed 17-5 / Cunningham Creek | 1 | 6.098 | 34.71 | Clark | 44.56 / -90.38 | Neillsville / Marshfield |
| Blake Creek | 2 | 11.184 | 52.54 | Waupaca | 44.52 / -89.01 | Waupaca |
| Magdanz / Hatton Creek | 3 | 6.911 | 43.25 | Waushara | 44.23 / -89.03 | Waupaca / Wautoma |
| Little Wolf River (North Branch) | 4 | 13.660 | 80.81 | Waupaca | 44.47 / -89.01 | Waupaca |
| Sucker Creek | 5 | 8.816 | 40.48 | Waushara | 43.99 / -89.18 | Wautoma |
| West Branch Shioc River | 6 | 6.338 | 41.44 | Shawano | 44.71 / -88.47 | Shawano |
| 2008 | | | | | | |
| Bronken Creek | 7 | 8.201 | 14.87 | Dunn | 45.05 / -91.76 | Bloomer / Ridgeland |
| Gillespie Creek | 8 | 4.130 | 23.24 | Polk | 45.70 / -92.26 | Cumberland / Grantsburg |
| Onemile Creek | 9 | 12.455 | 114.48 | Juneau | 43.76 / -90.09 | Mauston |
| Spring / South Branch Pine Creek | 10 | 11.825 | 92.20 | Dunn | 45.22 / -91.87 | Ridgeland |
| Walla Walla Creek | 11 | 15.019 | 61.13 | Waupaca | 44.28 / -89.03 | Waupaca |
| Webster Creek | 12 | 10.894 | 40.40 | Juneau | 43.84 / -90.19 | Mauston |

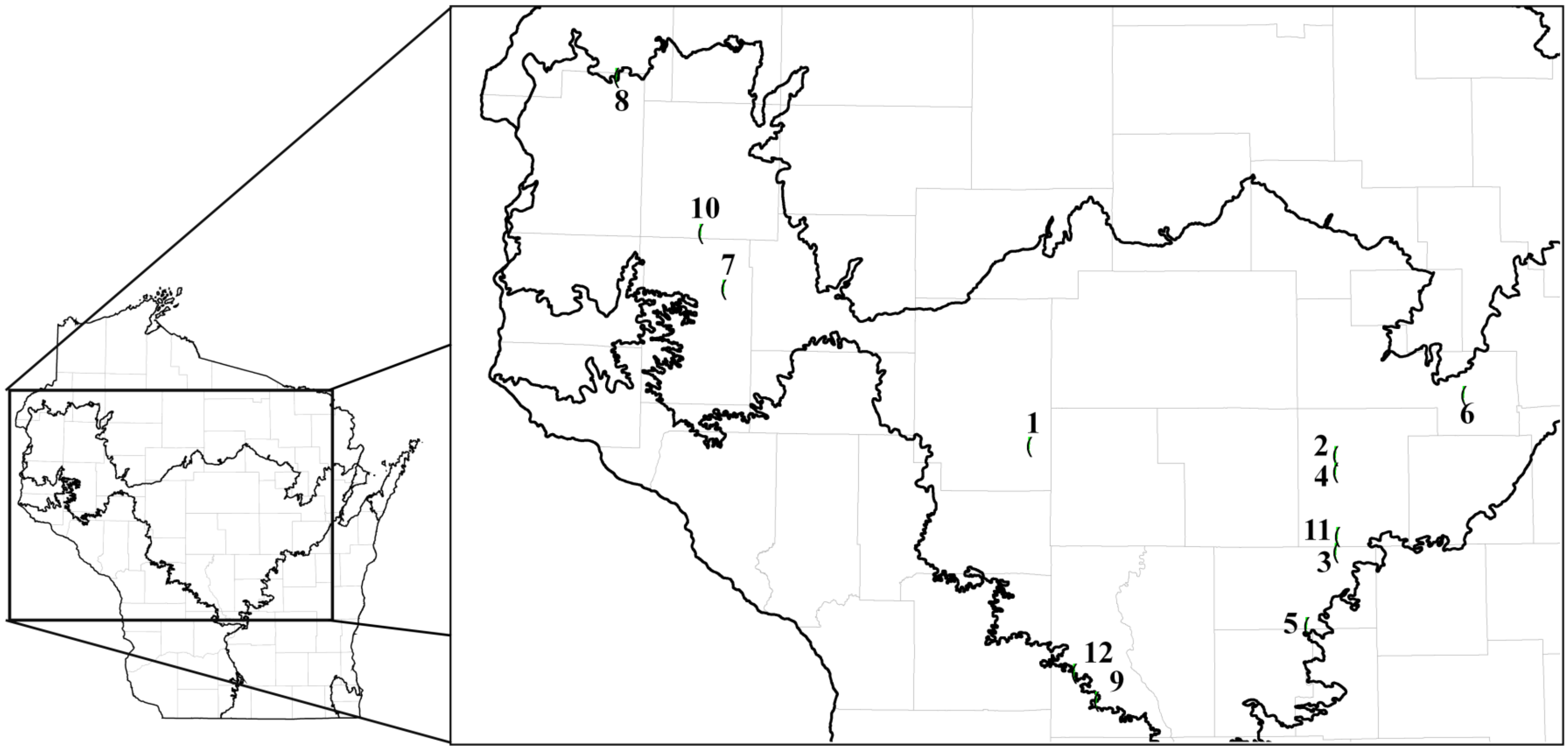



**Figure 3.** A map of study creek locations in Wisconsin’s North Central Hardwood Forest Ecoregion (Omernik and Gallant. 1988) with creek identification numbers (Table 2).

*Stream Segmentation*

Streams were broken into segments based on riparian vegetation type in order to relate stream temperature changes to vegetation types and characteristics that each vegetation type possessed which could influence thermal characteristics of streams (such as percent shading). Initial segments were delineated using Geographic Information Systems (GIS) and orthophotos and were later ground-truthed for accuracy using visual assessments. Vegetation types were divided into six main categories: 1) grass, 2) shrub (woody plants < 5 m tall), 3) tree (woody plants ≥ 5 m tall), 4) transitional grass, 5) transitional shrub (woody plants < 5 m tall), and 6) transitional tree (woody plants ≥ 5 m tall) (Table 3).

**Table 3.** Stream segment vegetation category classification criteria.

| Segment Classification | Criteria |
|---|---|
| (1) Grass | ≥ 75% Stream shading provided by grasses |
| (2) Shrub (< 5 m) | ≥ 75% Stream shading provided by shrubs (< 5 m) |
| (3) Tree (≥ 5 m) | ≥ 75% Stream shading provided by trees (≥ 5 m) |
| (4) Transitional Grass | Greatest percentage of stream shading from grasses but less than 75% |
| (5) Transitional Shrub (< 5 m) | Greatest percentage of stream shading from shrubs (< 5 m) but less than 75% |
| (6) Transitional Tree (≥ 5 m) | Greatest percentage of stream shading from trees (≥ 5 m) but less than 75% |

*Segment Transects*

Physical habitat within each stream segment was surveyed using cross-sectional transects. Transects were equally spaced relative to the length of each segment in order to obtain a minimum of five transects per segment. Segment transects were spaced by measuring the segment length to the nearest meter and dividing it by 5, then rounding down to the nearest number that was a product of 10; this number was the distance between the transects or interval distance. The first transect will occur at half the interval distance so as not to place transects at the very beginning or end of segments. If the segment was greater than 500 m and less than 1,000 m, then 100 m was the interval between transects. If the segment was equal to or greater than 1,000, meters then the interval distance between transects was 200 m. If transects occurred less than 5 meters away from the end of the segment they were not measured. For instance:

1) Segment length= 76 m
2) 76 / 5 = 15.2, then
3) Rounding down to the nearest product of 10, 10 m is the interval distance.
4) 10 / 2 = 5, the first transect is at the fifth meter of the segment
5) 5 + 10 = 15, the second transect is at the fifteenth meter of the segment, etc…

Using this method, the transects for a segment 76 m long will occur at 5, 15, 25, 35, 45, 55, and 65 m. This method was chosen as a way of randomly sampling the physical characteristics of each stream segment. Transects were defined as a line perpendicular to the flow of water, along which measurements are taken. Point measurements were taken at four, equally spaced intervals and one measurement in the thalweg resulting in five point measurements (Simonson et al. 1994). Measurements along transects were measured as close to summer baseflow (July, August) as possible.

*Stream Segment Morphology*

To characterize the stream segment morphology, measurements collected included stream width, depth, undercut banks, stream channel bank angle, entrenchment, flow, slope, and length (Table 4). Variables measured at either individual points (e.g., depth), quadrats (e.g., embeddedness and substrate type), or transects (e.g., stream width, buffer width, percent shade, percent riparian land use type) were averaged to assign an overall quantitative value to each segment for that variable. Morphologic features at sample point and transect scales were

**Table 4.** Measured stream habitat variables, descriptions, and the scale at which they were sampled. Forty-four habitat variables describing stream morphology, stream temperature, riparian vegetation, land use, shading factors, meteorological, and substrate characteristics are included. Each variable is indented when numerically coded in a category (modified from Platts et al. 1987; Rosgen 1994; Simonson et al. 1994; WI DNR 2002b; Bartholow 2004; WI DNR 2004).

| Variable | Description | Scale | Source |
|---|---|---|---|
| | **Stream Morphology** | | |
| Depth (m) | Vertical distance from the stream bed to the water surface | Point | Platts et al. (1987) |
| Entrenchment | Vertical containment of the river and the degree to which it is incised into the valley floor | Transect | WI DNR (2002b) |
| Discharge ($m^3/s$) | Volume of water per unit time traveling downstream | Site | WI DNR (2002b) |
| Segment length (m) | Length of segment taken using GIS techniques | Segment | Bartholow (2004) |
| Segment slope (m/m) | Elevation change divided by the segment length | Segment | WI DNR (2002b) |
| Stream channel bank (°) | Angle of bank directly adjacent to the water surface in degrees | Transect | Platts et al. (1987) |
| Stream width (m) | Wetted width of the stream at the water surface, from bank to bank | Transect | Platts et al. (1987) |
| Undercut bank (m) | Depth of bank overhanging the water | Transect | Simonson et al. (1994) |
| | **Riparian Vegetation, Land Use, and Shading Factors** | | |
| Buffer width (m) | Length of contiguous undisturbed land uses from the stream's edge out to 10 m | Transect | Simonson et al. (1994) |
| Riparian land use (%) | Description of land use and vegetation cover type within 5 m of the stream bank | Transect | Simonson et al. (1994) |
| (1) Wetland (%) | Land that is poorly drained and covered with standing water for much of the year, including swamps and marshes | | |
| (2) Meadow (%) | Land dominated by grass and forbs with few woody plants and not subject to regular mowing or grazing | | |
| (3) Shrub (%) | Land dominated by trees and woody vegetation generally < 3 m high | | |
| (4) Woodland (%) | Land dominated by trees > 3 m high | | |
| (5) Pasture (%) | Land, either wooded or grassy, regularly grazed by livestock | | |
| (6) Cropland (%) | Land plowed and planted with row-crops and harvested on a yearly basis, plus actively cultivated orchard and regularly mowed hayfields | | |
| (7) Barnyard (%) | Land used to confine and feed high densities of livestock | | |
| (8) Residential (%) | Land modified for human use, including buildings, roads, parking lots, and recreational grounds | | |
| (9) Bare soil (%) | Land covered by bare soil | | |
| (10) Other (%) | Land that cannot be included in the other categories | | |

**Table 4. (Continued)**

| Variable | Description | Scale | Source |
|---|---|---|---|
| | **Riparian Vegetation, Land Use, and Shading Factors** | | |
| Topographic shade (°) | Shade provided to the center of the stream by the banks and the stream valley in degrees | Transect | Bartholow (2004) |
| Shading (%) | A measure of the amount of shade the stream receives | Transect | Platts et al. (1987) |
| Segment elevation (m) | Average elevation of the stream for a segment | Segment | Bartholow (2004) |
| Segment latitude (°) | Angular distance north and south of the equator in degrees | Segment | Platts et al. (1987) |
| Segment orientation (°) | Degrees the stream deviates from a north to south orientation | Segment | Platts et al. (1987) |
| Vegetation crown (m) | Width of shade providing vegetation | Transect | Platts et al. (1987) |
| Vegetation offset (m) | Distance major vegetation type is located away from the stream | Transect | Platts et al. (1987) |
| Vegetation overhanging | Thick overhanging vegetation within 0.1 m of the water surface | Transect | Simonson et al. (1994) |
| Vegetation overstory | Height of vegetation providing shade to the stream | Transect | Platts et al. (1987) |
| | **Meteorological** | | |
| Air temperature (°C) | Temperature in degrees Celsius | Site | Bartholow (2004) |
| Possible sun (%) | Indirect and inverse measure of cloud cover | Site | Bartholow (2004) |
| Relative humidity (%) | Amount of water vapor in the air | Site | Bartholow (2004) |
| Wind speed (mph) | Speed of wind in miles per hour | Site | Bartholow (2004) |
| | **Substrate Characteristics** | | |
| Embeddedness (%) | The degree to which coarse substrate is surrounded or covered by fine sediment | Point | Simonson et al. (1994) |
| Substrate type (%) | Description of substrate type and prevalence | Point | Simonson et al. (1994) |
| (1) Silt (%) | Substrate of particles < 0.062 mm | | |
| (2) Sand (%) | Substrate of 0.062-2.0 mm particles | | |
| (3) Gravel (%) | Substrate of 2.0-63.5 mm particles | | |
| (4) Cobble (%) | Substrate of 63.5-254 mm particles | | |
| (5) Boulder (%) | Substrate of particles > 254 mm | | |
| | **Stream Temperature** | | |
| Water temperature (°C) | Temperature in degrees Celsius | Segment | WI DNR (2004) |

measured using a tape measure, measuring stick, and clinometer. Stream segment length and slope were measured at the segment scale using GIS. Stream width was measured as the wetted width of the stream surface, not including islands, exposed bars, backwater, and adjacent wetlands, measured from one bank to the other and perpendicular to the direction of the stream flow and recorded to the nearest 0.01 m (Simonson et al. 1994). Stream depth was measured at each point sample and averaged to obtain the mean depth for each transect. Undercut banks were defined as banks vertically overhanging the water surface by no more than 0.10 m, undercut by at least 0.20 m, and the water under the bank must be at least 0.20 m deep; these were measured to the nearest cm (Simonson et al. 1994). Stream channel bank angle is the angle of the bank at the water's edge. It was measured in degrees with a clinometer on each bank along each transect (Platts et al. 1987). Entrenchment, the ratio of the flood-prone area width to the bankfull surface width of the channel, was measured along a transect and provided a relative measure of how much the channel was incised (Rosgen 1994). Discharge was measured at the beginning and end of each study stream, at the mouth of tributaries, in the study stream just above the confluence with tributaries, and at various easily accessible locations throughout. Discharge was measured in $m^3/s$ using the technique described in WI DNR (2002a) while not allowing more than 10% of the total flow to go through any one cell. This was done in order to obtain more accurate flow measurements required for modeling small stream temperatures. Stream slope was measured at the segment scale using GIS and 1:24,000 USGS topographic maps by dividing the elevation change of the stream segment by the segment length (Rosgen 1994).

*Substrate Characteristics*

In order to quantify the substrate characteristics, embeddedness and substrate type were visually estimated at the point scale using a 0.3 m square quadrat centered over transect points (Table 4). Embeddedness is the extent to which course substrate is surrounded or covered by fine sediment. It was estimated on a scale of 0, 25, 50, 75, and 100% with 100% meaning totally buried and 0% meaning not covered or surrounded at all. Substrate type was visually estimated to the nearest 5% in each quadrat.

*Stream Temperature*

Stream temperature was measured at the beginning and end of each study stream's surveyed length in the summer of 2007 with Onsest HOBO® Water Temp Pro v2 temperature data loggers recording continuously at 30 minute intervals (Table 4). In 2008, stream temperature was measured at the upstream and downstream end of each stream segment with intervals not exceeding 1,000 m which included the beginning and end of the surveyed length of each stream. This was done in order to obtain an accurate temperature profile for each stream, relate stream temperature change to each segment's characteristics, and validate the stream temperature model predictions at the end of each segment. Due to field segment-length measurement errors and the loss of several temperature loggers, the interval between loggers was as much as 1,953 m on one stream (e.g., West Branch Shioc River). Data loggers were also placed in tributaries feeding the study stream just far enough upstream to avoid any mixing with the study stream. Data loggers were placed in the streams prior to June 1 and were taken out after August 31 to insure that maximum summer stream temperatures were recorded. In the summer of 2007, temperature data loggers were placed starting on June 1 and were not all placed until June 20; this did not affect the measurement of MWAT or MDAT. All temperature loggers were placed in shaded areas with well mixed water that remained wet in the driest conditions and were free of sediment to ensure representative water temperature readings. Prior to field use, each temperature logger was checked for accuracy by activating them, submerging them in an ice bath and then comparing the recorded temperatures to the actual water temperature as manually measured by a thermometer with an accuracy of 0.2°C (WI DNR 2004).

*Riparian Vegetation, Land Use, and Shading Factors*

Riparian vegetation, land use, and shading factors including buffer width, land use, topographic altitude, shading, segment elevation, segment latitude, segment orientation, vegetation crown, vegetation offset, vegetation overhang, and vegetation overstory height were measured for modeling purposes (Table 4). Buffer width was measured from the stream edge to 10 m upland and land use percentages were visually estimated out to 5 m from the stream edge along the transect line (WI DNR 2002b). Topographic shade (as opposed to vegetation shading) is a measure of the amount of shade derived from stream banks and the valley that a stream receives and is measured from the center of the stream along the transect line in degrees using a clinometer. Vegetative shading is measured in percent using a modified spherical densiometer according to Platts et al. (1987) along the transect line and is then averaged using a technique modified from Lawson et al. (2005) in order to be based on a 100% scale. This technique averages the number of shaded measurement points on a modified concave spherical densitometer for four measurement locations across the transect line. These four measurements were taken along the transect line 30.5 cm (12 inches) above the water at the middle of the stream facing upstream and downstream as well as 30.5 cm perpendicular to the left streambank facing the left bank and 30.5 perpendicular to the right bank facing the right bank. Elevation and latitude are all average values calculated for each segment. Elevation and latitude were measured using GIS, global positioning system (GPS), and topographic maps. Segment orientation represents the degrees a stream deviates from a north to south orientation taken with a compass at each transect (Platts et al. 1987). Vegetation crown, offset, and overstory height measurements are all measured in meters to the nearest cm taken from the major shade providing plant on both banks at each transect (Platts et al. 1987). Vegetation crown is the maximum diameter of the crown for all hardwoods, the maximum radius for softwoods to account for the tapering crown softwoods possess, and is considered to be 0.02 m for all grasses. Offset is the distance the center of the plant is away from the edge of the stream on the transect line. Overstory height is the height of the plant in meters. Vegetation overhanging is the depth of thick overhanging vegetation within 0.1 m of the water surface measured to the nearest cm on both banks at every transect if it occurs (Simonson et al. 1994).

*Meteorological*

Several meteorological variables were recorded including air temperature, percent possible sun, relative humidity, and wind speed (Table 4). Meteorological variables were averaged for each modeling period. Air temperature was recorded from the closest weather station or by averaging values when weather stations occurred at similar distances from the study stream (Table 2). Possible sun is the indirect and inverse measure of cloud cover measured in a percent. Relative humidity is the amount of water vapor found mixed in the air also measured in a percent. Possible sun, relative humidity, and wind speed were obtained from local climatological data (LCD) reports from major airport weather stations because these data are not reported by smaller local weather stations (Bartholow 2004). The closest weather station to a study stream reporting LCD was used for recording relative humidity and wind speed. These distances ranged from 37.5 to 120.1 km away. Percent possible sun was only recorded at the

Green Bay, WI International Airport and was used for all study stream temperature modeling. The distance from Green Bay, WI International Airport to study streams ranged from 37.5 to 353.6 km. Mohseni et al. (1998) found no significant effect from the distances of weather stations to stream locations ranging from 1.4 to 244 km on the goodness of fit of their nonlinear regression model for weekly stream temperatures throughout the United States. Since a weekly average was used for modeling, an acceptable difference between study stream meteorological conditions and weather recording locations was assumed.

## Data Analysis

### *Temporal Thermal Profiles*

Thermal profiles for all study streams were created for the modeling period, when maximum weekly average temperature (MWAT) occurred. These profiles were created using the recorded stream temperature data from the Onsest HOBO® Water Temp Pro v2 temperature data loggers recording continuously at 30 minute intervals.

### *Heat Budget Model*

The Stream Segment Temperature Model (SSTEMP) was chosen as the heat budget model for this analysis (Bartholow 1989). This model is a steady-state physics-based model that includes stream geometry, hydrology, meteorology, and shade inputs that are averaged for each modeling segment making this a steady-state model. The model computes the heat exchange of an amount of water (volume) from an initial inflow water temperature throughout a segment by simulating heat flux processes including atmospheric radiation, convection, conduction, evaporation, solar radiation, stream back radiation, and vegetative radiation to calculate a downstream water temperature.

Several assumptions are made when using SSTEMP. One assumption is that all the water in the system is immediately and completely mixed which means there are no vertical or lateral temperature gradients in the streams as have been shown to occur in some large pools and areas adjacent to the thalweg (Bartholow 1989). However, most streams in this study were small with relatively few large pools and thus mixing was most likely was not a problem. It is also assumed that all the model variables can be appropriately described by a mean value for the time period even though stream width, shade, and other variables are likely to vary to a certain degree throughout a segment. Lateral inflow from groundwater is also assumed to be distributed evenly throughout the segment. This is likely one of the areas where some of the greatest stream temperature modeling errors could occur. Groundwater upwelling (both distribution and amount) has been shown to largely influence stream temperatures but is highly variable and difficult to measure exactly (Gaffield et al. 2005). Modeling time period, MWAT, is assumed to be sufficiently long enough to allow water to flow through the entire length of a segment so as the water can be completely exposed to the conditions of the stream segment. The last assumption is that SSTEMP does not account for cumulative effects. This means that when one variable is modified that would likely affect another variable; the effected variable does not change unless it is manually done (Bartholow 2000 and Bartholow 2004).

SSTEMP was run according to Bartholow (2004) using similar input variables (Table 5). The model was executed using the “process external file” option where input variables were placed into a Microsoft Excel® spreadsheet and saved in comma-separated values (CSV) format; output values are then saved in a CSV file created by SSTEMP. SSTEMP outputs include predicted mean and mean equilibrium stream temperatures for the modeling period and are

**Table 5.** Input variable requirements of SSTEMP (Bartholow 2004).

| | Variable | Data Source |
|---|---|---|
| **Hydrology** | | |
| | Inflow temperature (°C) | Temperature data loggers / calculated by SSTEMP |
| | Segment inflow (cfs) | Field measurement |
| | Segment outflow (cfs) | Field measurement |
| | Accretion temperature (°C) | Local weather station |
| **Geometry** | | |
| | Latitude (%) | Geographic Information System |
| | Dam at head of segment (present/absent) | No dams in any streams |
| | Segment length (km) | Geographic Information System |
| | Upstream elevation (m) | Geographic Information System |
| | Downstream elevation (m) | Geographic Information System |
| | Width's A term (sec/m$^2$) | Calculated from field measurements |
| | Width's B term (dimensionless) | SSTEMP default value (0.2) |
| **Time of Year** | | |
| | Middle day of modeling period (mm/dd) | Temperature data loggers |
| **Meteorology** | | |
| | Air temperature (°C) | Local weather station |
| | Relative humidity (%) | Local climatological data report |
| | Wind speed (mi/hour) | Local climatological data report |
| | Ground temperature (°C) | Local weather station |
| | Thermal gradient (J/m$^2$/sec/°C) | SSTEMP default value (1.65 J/m$^2$/sec/°C) |
| | Possible sun (%) | Local climatological data report |
| | Dust coefficient (dimensionless) | SSTEMP default value (6) |
| | Ground reflectivity (%) | SSTEMP default value (20%) |
| | Solar radiation (J/m$^2$/sec) | Calculated by SSTEMP |
| **Shade** | | |
| | Total shade (%) | Field measurement |

reported in the results. Theurer et al. (1984) and Bartholow (1989) describe the complete model in detail.

Models for all study streams during the period of maximum weekly average temperatures (MWAT) were created, calibrated, and verified to best represent the actual recorded stream temperatures. Stream temperatures were modeled from the location of the temperature data logger furthest upstream with the exception of Unnamed Creek 17-5 in which modeling began at the data logger second farthest upstream since recorded flows were too low, 0.0789 cfs, for the model to accept at the initial data logger location. SSTEMP was then calibrated for each stream individually specifically for the time period of MWAT. Calibration was carried out by attuning flow, accretion temperature, width, and shading variables appropriately (Bartholow 1989; Bartholow 2004). Flow adjustments were the primary form of calibration because specific measurements were not made at each segment's starting and ending point. These calibrations were performed by adjusting the inflow and outflow discharges in a given segment. All stream models were successfully calibrated and verified except for Gillespie Creek (Appendix A and B). Gillespie Creek's actual compared to modeled stream temperatures were on average 1.69°C cooler. There was no logical calibration that could be carried out for modeling this stream. Since cooler stream temperatures could not be predicted by the heat budget equation, it is likely that a cooling factor was not present in the model and therefore could not account for this cooling. It is the belief of the authors that this cooling was due to groundwater and hyporheic flow exchanges. Linear regression between the upstream and downstream recorded stream temperatures was used to calculate the models' deviation from actual temperatures when temperature data loggers were not present at the end of a segment.

Once calibrated, modeled stream temperatures deviated from actual recorded temperatures by 0 to 1.24°C per segment on all twelve study streams with an average deviation of 0.09°C. Magdanz / Hatton Creek model was calibrated with little alterations in flow, but the last segment's accreation temperature was calibrated to be 7.05°C higher than estimated groundwater input temperatures because the water entering the channel was actually surface water springs that flowed through an open meadow before entering the stream. On Unnamed 17-5 / Cunningham Creek, one modeling point was an outlier and deviated from the regression between temperature measurement points by 2.4°C (Appendix B). This value was excluded because there was no temperature data logger near the end of that segment and it went through the extreme condition of a feed lot with no shade. SSTEMP was deemed accurate once calibrated for this subset of streams.

Calibrated models were used to predict how various levels of shading would affect temperature profiles; shading levels simulated were 0, 25, 50, and 75%. At the end of each simulation, the distance of stream that would be thermally suitable for brook trout, using the thermal criteria set previously at 22.3°C for the MWAT (Eaton et al.1995) for each study stream was recorded. Modeled increases or decreases in stream length suitable to brook trout were assessed to view the potential gains/losses from various stream shading levels when compared to actual stream temperature profiles. When streams were modeled and deemed thermally suitable for brook trout at the downstream end of the study segment (where flow and width measurements were taken), the predicted stream equilibrium temperatures were reported (i.e., the stream was deemed stable and at equilibrium; still being suitable at the end of the segment).

*Empirical Stream Temperature Model*

Downstream stream segment temperatures were predicted using multiple regression techniques. All stream temperature modeling was conducted during the period when maximum weekly average temperatures (MWAT) occurred. Stream segments with recorded upstream and downstream temperatures from 2007 and 2008 were used to create these models (n=134). Forty-six variables summarizing stream segments were used to create these models and residuals were checked for normal distribution (Appendix C). Natural log transformations were conducted on several variables to reduce deviations from normal distribution of residual error. To reduce effects of colinearity, models were created using stepwise multiple regression techniques in SAS (SAS Institute 2001) with alpha=0.05 to determine insertion of a variable into the model and alpha greater than 0.10 for variable removal. Residual sum of squares (SSE), adjusted $R^2$, corrected Akaike's Information Criteria ($AIC_c$), and Schwarz Bayesian Criteria (SBC) values were assessed to facilitate final model selection (Appendix E). Models were selected using $AIC_c$, however, SBC values have been found to correctly select the true model most consistently and are also presented (Beal 2007). Using $AIC_c$ values to select the best model, the model with the lowest value is chosen. Models within two units of the lowest $AIC_c$, value are considered to be alternate models. Models within four $AIC_c$, units have been shown to provide empirical support for being the correct model (Burnham and Anderson 1998).

*Riparian Vegetation and Stream Temperature Responses*

To assess how differences in vegetation type affect changes in stream temperature per segment, analysis of covariance (ANCOVA) was used. Stream segments located throughout central Wisconsin were analyzed (n=59). Vegetation types (grasses, shrubs, and trees) were the main effects during the MWAT and MDAT periods. Upstream temperature was used as a covariate to account for the effects of upstream temperature has on overall stream temperature change. The dependent variables in the ANCOVAs were the change in stream temperature per kilometer ($\Delta$°C/km) and downstream temperature for the modeled time period. ANCOVA analyses were performed using SPSS (SPSS 2007). Upstream temperature, downstream temperature, and change in stream temperature per kilometer were tested for normality using a Kolmogorov-Smirnov one sample test and differences in variance using a Levene's test. Homogeneity of regression slopes for each factor was also tested. All assumptions of the ANCOVAs (equal variance, normal distribution, and homogeneity of regression slopes) were met and the ANCOVAs were run. Estimated marginal means were reported for each vegetation type. Alpha was set at $P \leq 0.05$ to detect significant differences between vegetation types. Bonferroni's multiple comparison method was used to compare riparian vegetation type effects on stream temperature change in order to define which vegetation types were significantly different if any ($P \leq 0.05$). Parameter estimates were used to construct graphs showing the effects of vegetation type on stream temperature change per km provided a given upstream temperature. Graphs of the estimated marginal means were also displayed representing the change in temperature per km at the mean upstream temperature.

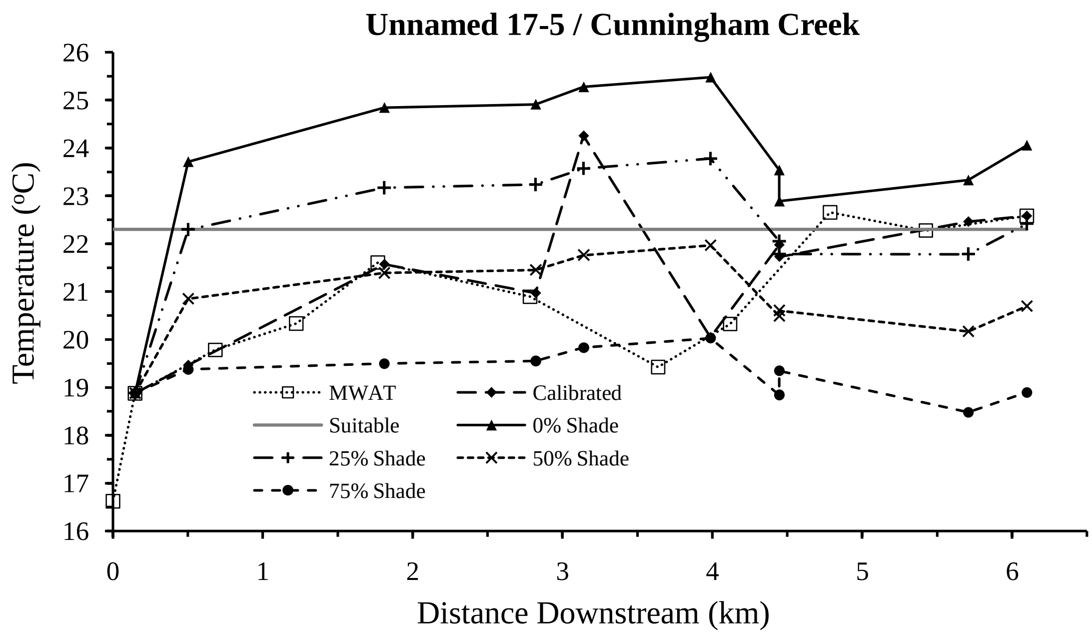
## RESULTS

Temporal Thermal Profiles

In 2007, water temperatures in six streams were monitored (Table 6). Study stream lengths ranged from 6.098 km on Unnamed 17-5 / Hatton Creek to 13.66 km on Little Wolf River (North Branch) with the length of thermally suitable brook trout water ranging from 3.856 km on Sucker Creek to the entire length of Blake Creek monitored, 11.14 km (Figures 4, 5, and 6; Table 6). Six additional stream course temperatures were monitored in 2008, with the length of stream monitored ranging from 4.13 km on Gillespie Creek to 15.019 km on Walla Walla Creek (Figures 7, 8, and 9; Table 6). The entire lengths of all the streams monitored in 2008 were later found to be thermally suitable for brook trout during the MWAT period. In fact, of the twelve streams monitored over the two summers only five crossed the thermal suitability threshold. Streams from both study years had a mean temperature of 18.8°C and ranged from 13.3 to 23.6°C during the MWAT period.

Heat Budget Model

Initially, the SSTEMP model was used to predict the stream temperatures for all stream segments in each study stream using the actual data. The SSTEMP model did a reasonable job overall at predicting the actual stream temperatures measured during the MWAT sample period as predicted stream temperatures followed the pattern of actual stream temperature profiles quite well (Appendix B). However, the predicted temperatures deviated substantially at a few segments, off by as much as 3.2°C on Spring / South Branch Pine Creek. Streams had an average maximum deviation from non-calibrated model predictions to actual stream temperatures of 2.15°C.



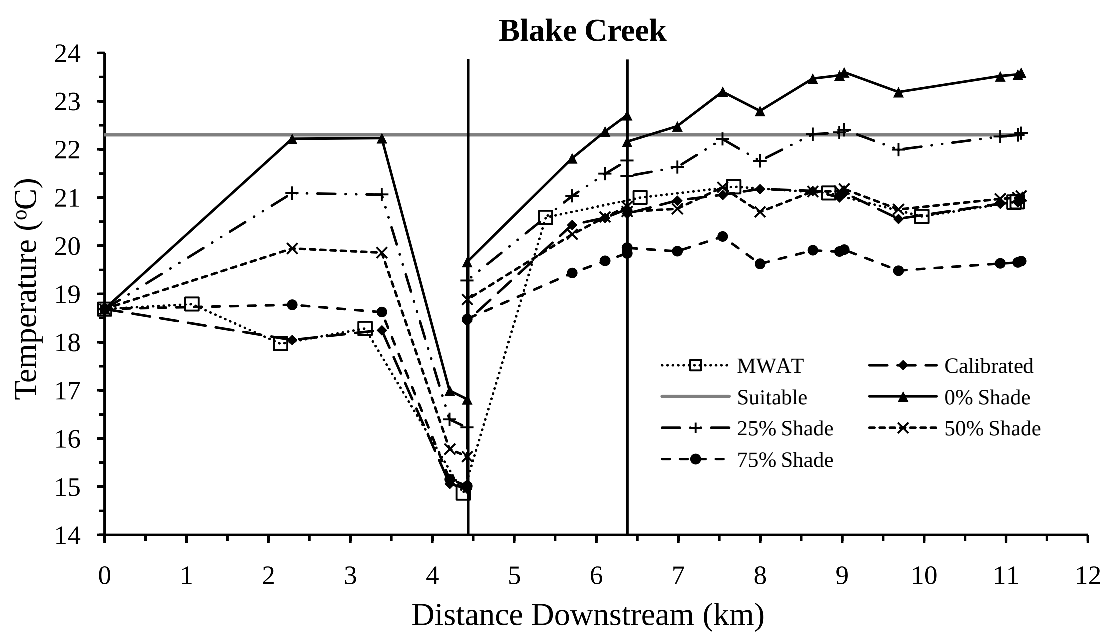
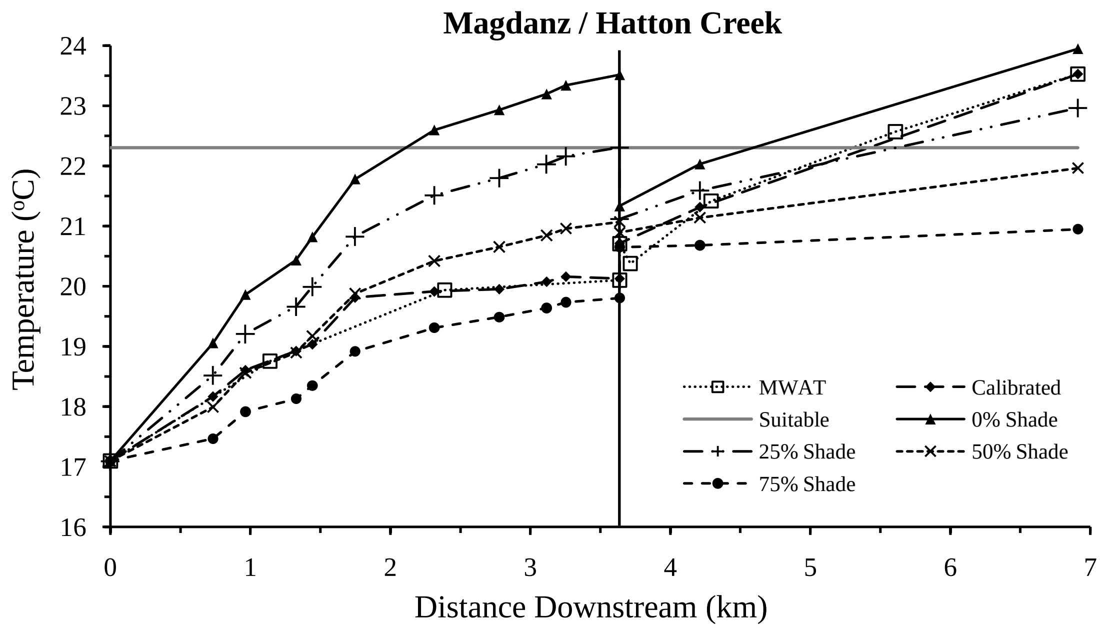




**Figure 4.** Thermal profiles of Unnamed 17-5/Cunningham Creek and Blake Creek during the period of MWAT in 2007 compared to modeled stream temperatures under various levels of shading with the suitability threshold for brook trout set at 22.3°C. Vertical lines represent locations of tributaries entering the streams.



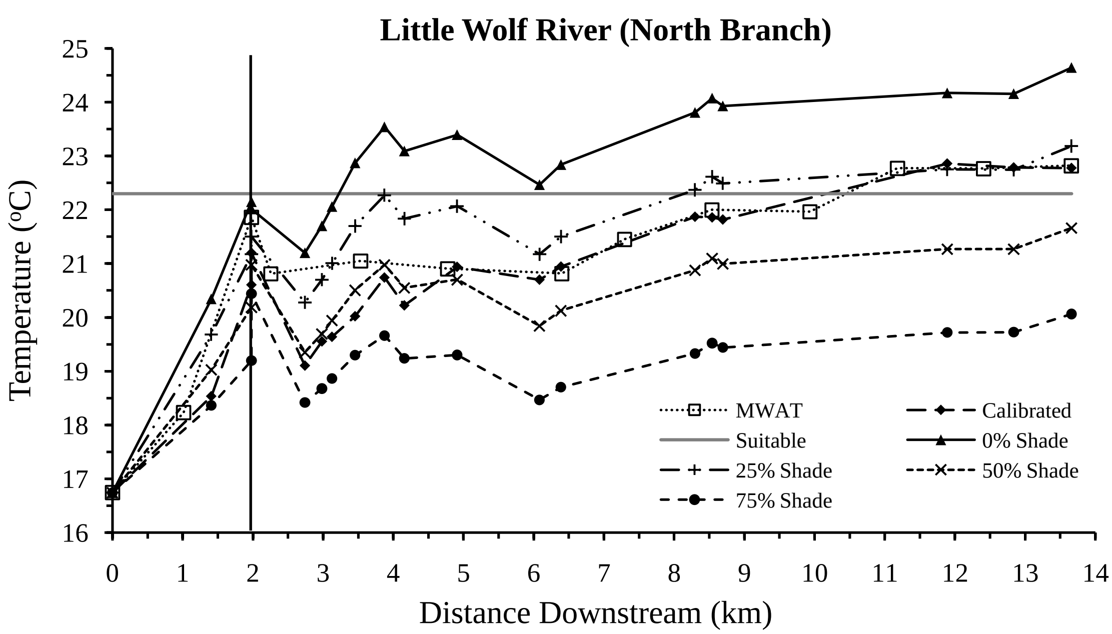
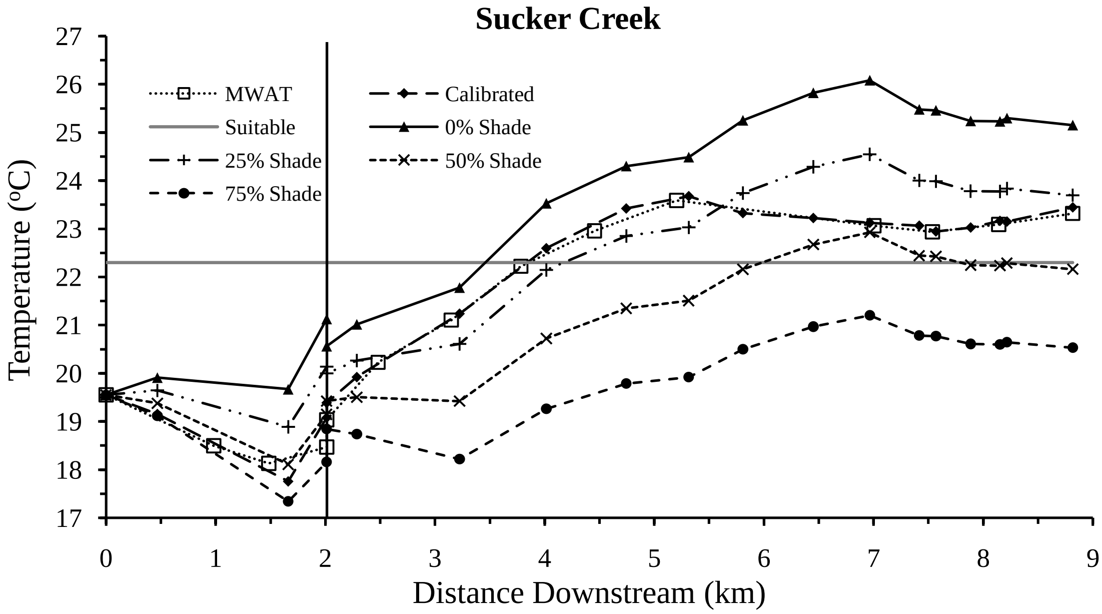




**Figure 5.** Thermal profiles of Magdanz / Hatton Creek and Little Wolf River (North Branch) during the period of MWAT in 2007 compared to modeled stream temperatures under various levels of shading with the suitability threshold for brook trout set at 22.3°C. Vertical lines represent locations of tributaries entering the streams.



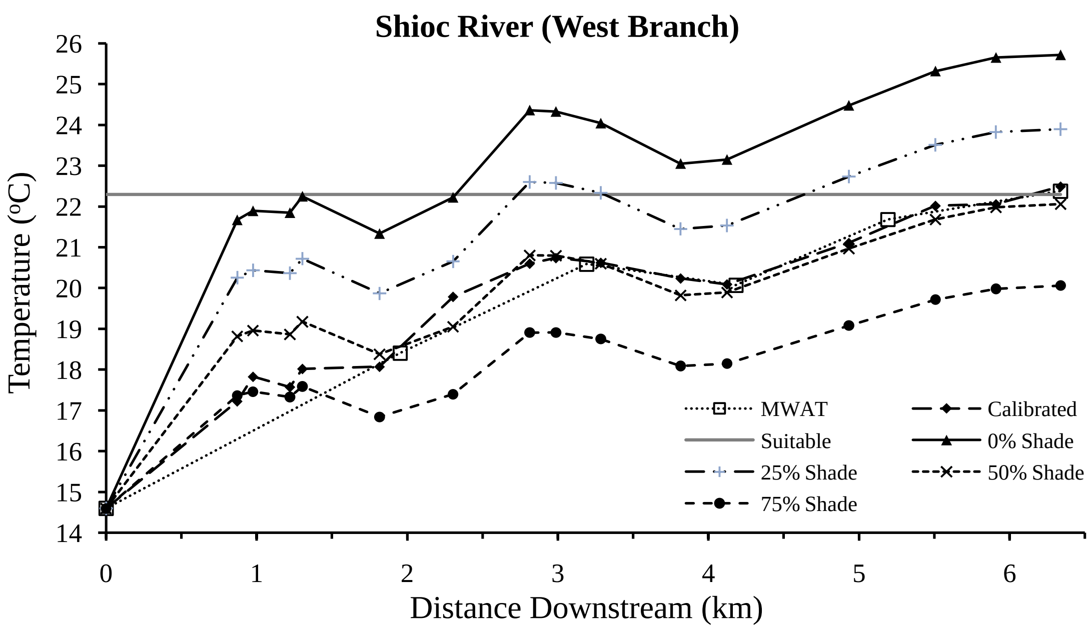
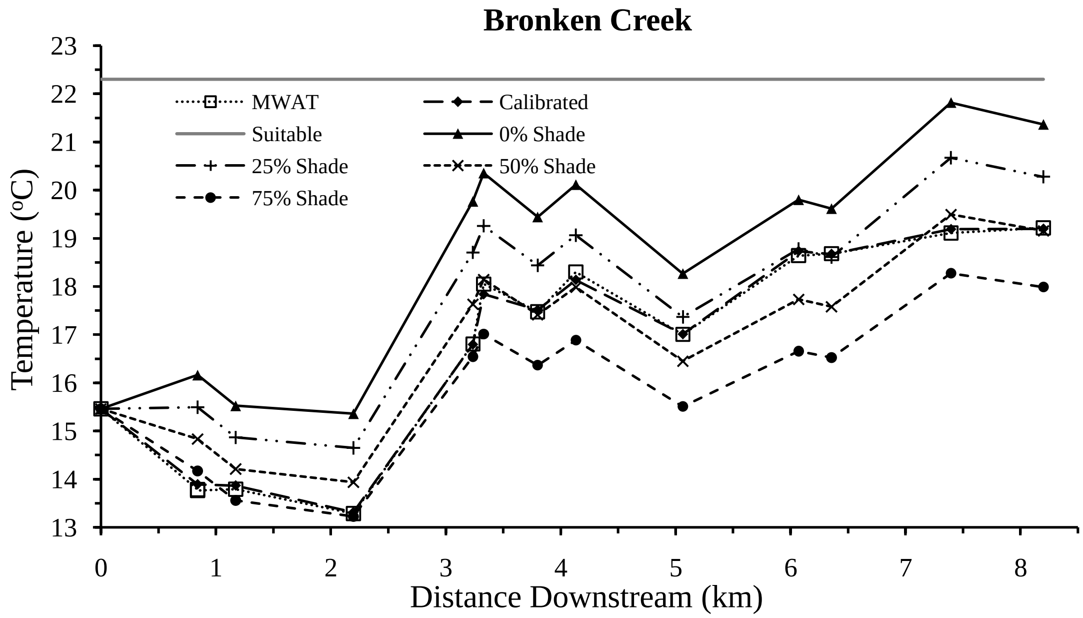




**Figure 6.** Thermal profiles of Sucker Creek and Shioc River (West Branch) during the period of MWAT in 2007 compared to modeled stream temperatures under various levels of shading with the suitability threshold for brook trout set at 22.3°C. Vertical lines represent locations of tributaries entering the streams.



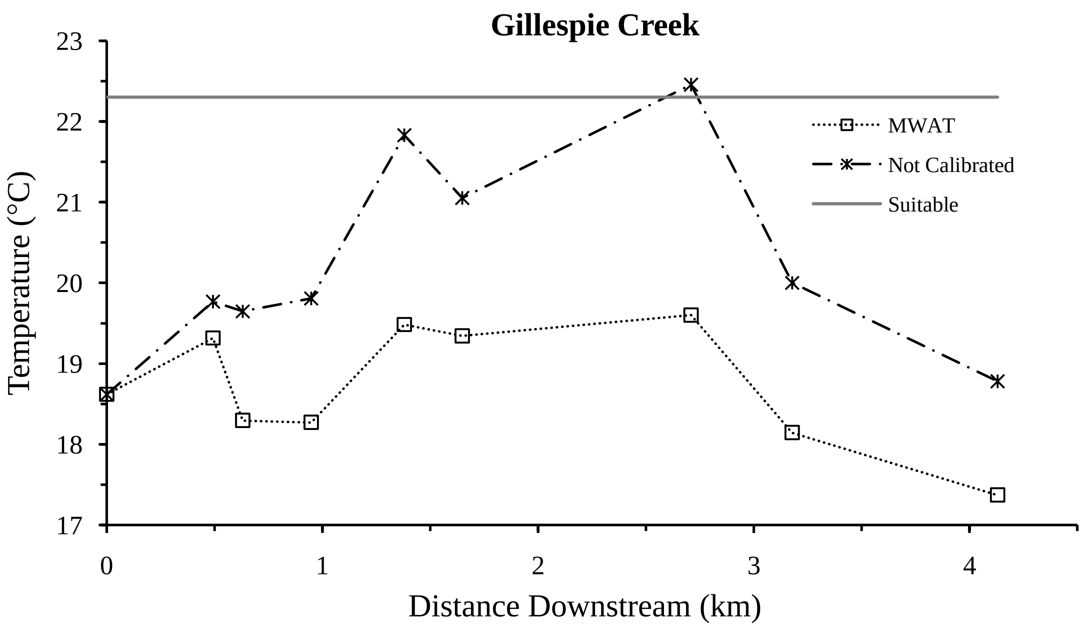
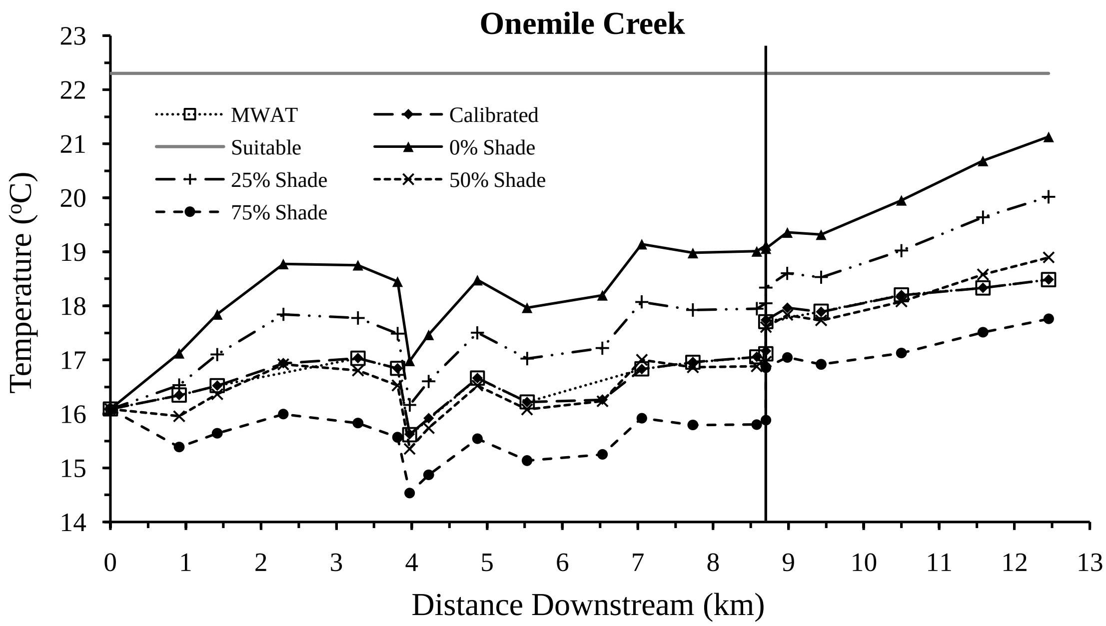




**Figure 7.** Thermal profiles of Bronken Creek and Gillespie Creek during the period of MWAT in 2008 compared to modeled stream temperatures under various levels of shading with the suitability threshold for brook trout set at 22.3°C. Vertical lines represent locations of tributaries entering the streams. Gillespie Creek was not calibrated because no reasonable calibration could be made.



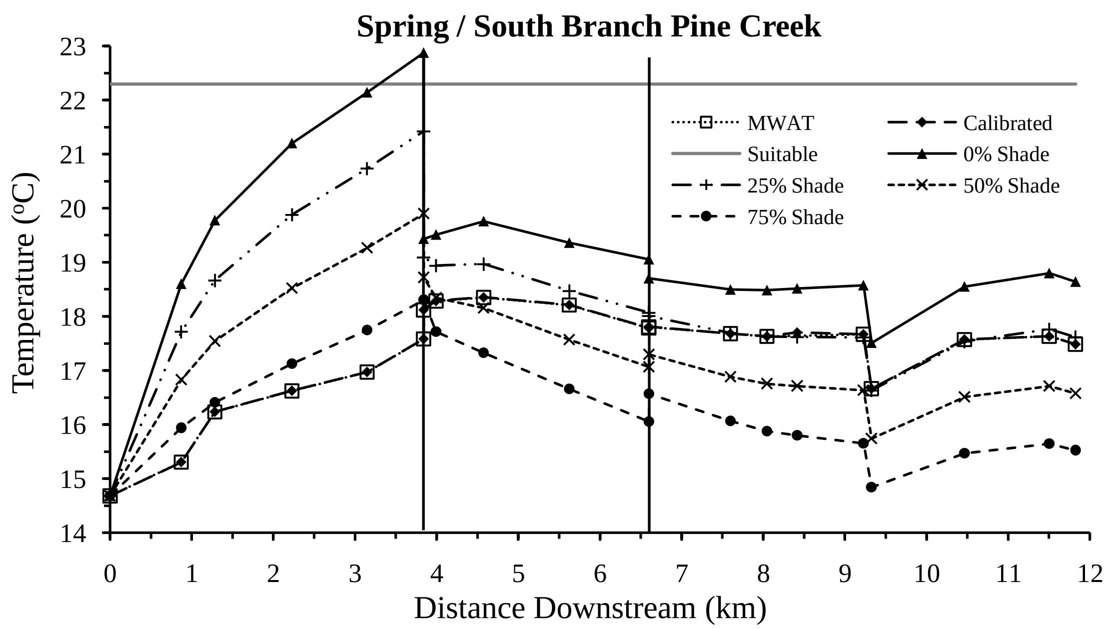
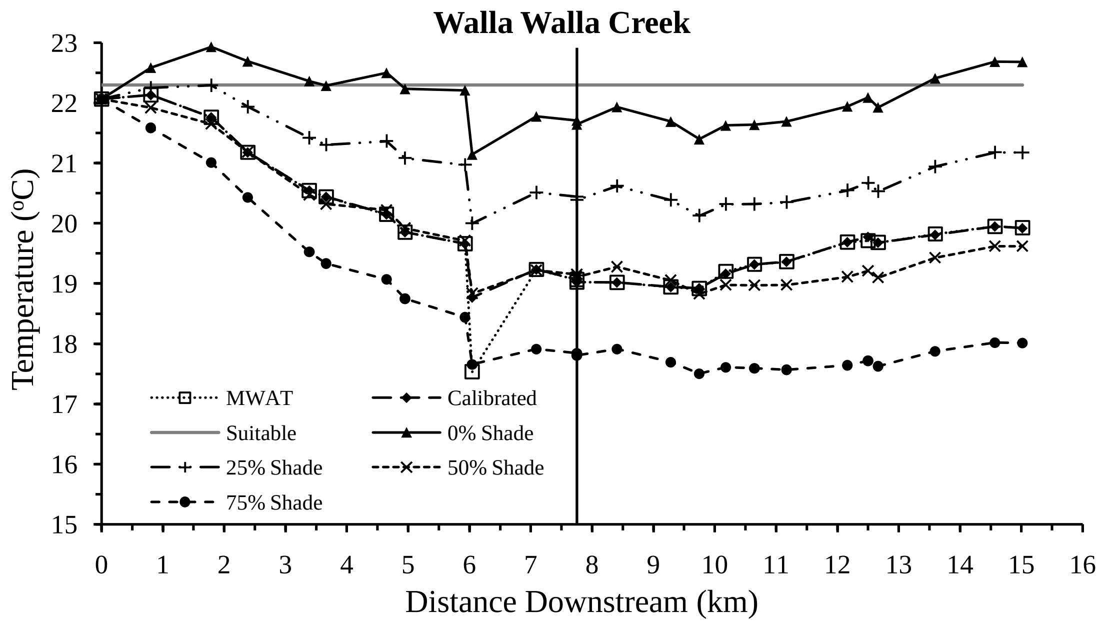




**Figure 8.** Thermal profiles of Onemile Creek and Spring / South Branch Pine Creek during the period of MWAT in 2008 compared to modeled stream temperatures under various levels of shading with the suitability threshold for brook trout set at 22.3°C. Vertical lines represent locations of tributaries entering the streams.



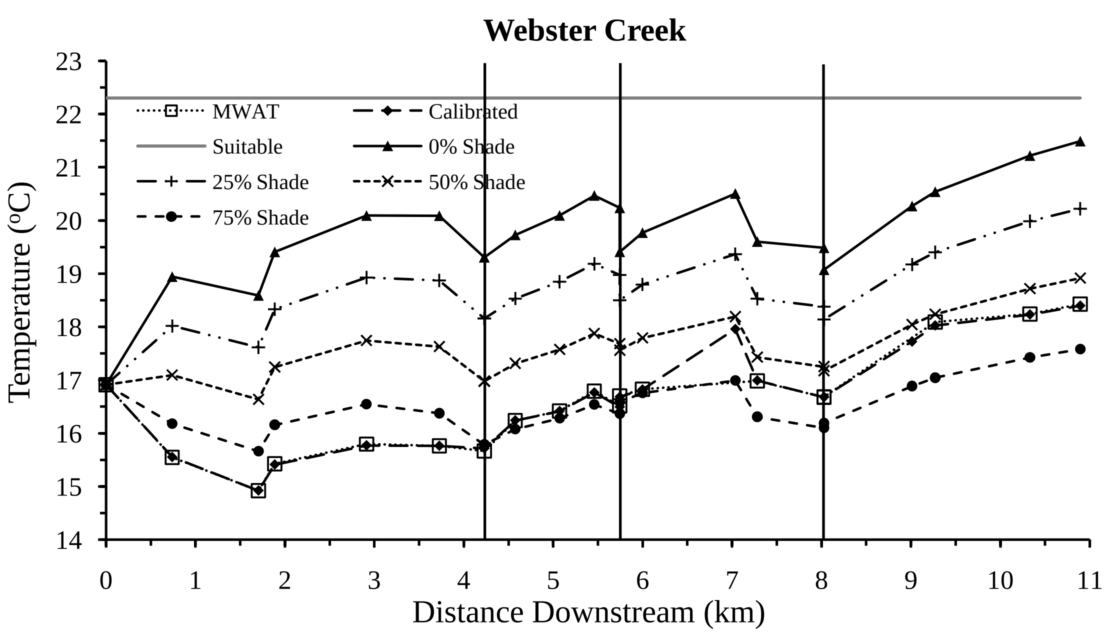




**Figure 9.** Thermal profiles of Walla Walla Creek and Webster Creek during the period of MWAT in 2008 compared to modeled stream temperatures under various levels of shading with the suitability threshold for brook trout set at 22.3°C. Vertical lines represent locations of tributaries entering the streams.

**Table 6.** Summary of calibrated SSTEMP modeling results from all twelve study streams compared to recorded stream temperatures and current lengths of stream suitable to brook trout during the period of maximum weekly average temperatures (MWAT).

| | Actual MWAT | | | Modeled MWAT | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Surveyed | | | 0% Shade | | | 25% Shade | | | 50% Shade | | | 75% Shade |
| Stream | Length (km) | Suitable Length (km) | % Length Suitable | Suitable Length (km) | % Increase | T_eq (°C) | Suitable Length (km) | % Increase | T_eq (°C) | Suitable Length (km) | % Increase | T_eq (°C) | T_eq (°C) |
| 2007 | | | | | | | | | | | | | |
| Unnamed 17-5 / Cunningham Creek | 6.098 | 4.706 | 77.17% | 0.398 | -91.54% | 26.08 | 2.149 | -54.33% | 24.36 | 6.098 | 29.58% | 22.52 | 20.54 |
| Blake Creek | 11.184 | 11.184 | 100.00% | 6.053 | -45.88% | 25.68 | 10.582 | -5.38% | 24.34 | 11.184 | 0.00% | 22.92 | 21.41 |
| Magdanz / Hatton Creek | 6.911 | 5.299 | 76.67% | 3.062 | -42.22% | 25.96 | 5.608 | 5.84% | 23.24 | 6.911 | 30.42% | 23.24 | 21.76 |
| Little Wolf River (North Branch) | 13.660 | 10.450 | 76.50% | 3.223 | -69.16% | 26.39 | 8.139 | -22.11% | 24.87 | 13.660 | 30.72% | 23.25 | 21.52 |
| Sucker Creek | 8.816 | 3.856 | 43.74% | 3.459 | -10.30% | 26.52 | 4.174 | 8.24% | 25.02 | 5.987 | 55.26% | 23.42 | 21.71 |
| Shioc River (West Branch) | 6.338 | 6.207 | 97.93% | 2.322 | -62.59% | 26.98 | 4.063 | -34.53% | 23.35 | 7.636 | 23.02% | 23.35 | 21.33 |
| 2008 | | | | | | | | | | | | | |
| Bronken Creek | 8.201 | 8.201 | 100.00% | 8.201 | 0.00% | 23.15 | 8.201 | 0.00% | 22.02 | 8.201 | 0.00% | 20.82 | 19.57 |
| Gillespie Creek | 4.130 | 4.130 | 100.00% | - | - | - | - | - | - | - | - | - | - |
| Onemile Creek | 12.455 | 12.455 | 100.00% | 12.455 | 0.00% | 28.31 | 12.455 | 0.00% | 26.82 | 12.455 | 0.00% | 25.23 | 23.53 |
| Spring / South Branch Pine Creek | 11.825 | 11.825 | 100.00% | 11.825 | 0.00% | 23.41 | 11.825 | 0.00% | 21.97 | 11.825 | 0.00% | 20.45 | 18.83 |
| Walla Walla Creek | 15.019 | 15.019 | 100.00% | 8.954 | -40.38% | 24.82 | 15.019 | 0.00% | 23.13 | 15.019 | 0.00% | 21.32 | 19.38 |
| Webster Creek | 10.894 | 10.894 | 100.00% | 10.894 | 0.00% | 25.47 | 10.894 | 0.00% | 23.90 | 10.894 | 0.00% | 22.24 | 20.46 |

Riparian shading vegetation affected stream temperatures (Figures 4, 5, 6, 7, 8, and 9; Table 6). Shading modeled at the 0% level led to a decrease in the length of stream thermally suitable to brook trout in seven of the eleven streams. Unnamed 17-5 / Cunningham Creek had the largest loss of trout water, predicted to decrease by 91.54%. When 25% shade was modeled, four of the eleven streams were predicted to have decreases in the length of suitable brook trout water with Unnamed 17-5 / Cunningham Creek predicted to decrease by 53.44%. Two of the eleven streams were predicted to have slight increases in suitable brook trout water by 5.84% on Magdanz / Hatton Creek and 8.24% on Sucker Creek. Conversely, 50% shading of streams resulted in increases in the length of suitable water in five of the eleven streams by as much as 55.26% on Sucker Creek and 3.21 km on the Little Wolf River (North Branch); no decreases were predicted.

Progressive increases in shading led to progressive decreases in equilibrium temperatures lengthening the amount of surveyed stream suitable to trout by as much as 128.63%, 4.96 km, on Sucker Creek when 75% shading was modeled. None of the eleven study streams modeled had equilibrium temperatures less than 22.3°C when 0% shade was modeled. When 25% shade was modeled two streams' equilibrium temperatures were lower than the suitability threshold, with 50% shade four streams, and with 75% shade modeled ten of the eleven study streams had equilibrium temperatures less than 22.3°C. The single stream above the threshold, Onemile Creek, had an equilibrium temperature of 23.53°C (Table 6).

<u>Empirical Stream Temperature Model</u>

Riparian and other variables clearly affect stream temperatures as shown by multiple regression analysis of a various streams throughout central Wisconsin. Segment lengths ranged from 40 to 1953 m long (mean = 751 m ±31.61se) (Appendix D). Slopes of these segments ranged from 0.00057 to 0.00579 m/m (mean = 0.00159 m/m ±0.00009 se). Average stream flow in a segment ranged from 0.00285 to 0.43381 cms with a mean flow of 0.10764 cms. Segment average wetted widths ranging from 1.06 m to as much as 13.42 on one occasion and had a mean width of 3.50 m.

Variables that influenced the prediction of downstream temperature included: upstream temperature, Δ flow/distance (cms/m), slope (m/m), sand substrate (%), shade (%), segment length (m), width to depth ratio (m:m), wetted width (m), and upstream flow (cms) (Table 7). Upstream temperature was the first variable in all the models due to its high correlation with downstream temperature (Appendix E). Final model selection was based on $AIC_c$ values and produced a seven variable model with its $AIC_c$ value being the lowest at -109.33. Variables loading into this model were upstream temperature (+), Δ flow/distance (-), slope (-), shade (-), segment length (+), wetted width (+), and upstream flow (-). The variables loading into model 8 were more significant (lower P-values) compared to the variables loading into model 7. The variables loading into model 7 included upstream temperature (+), Δ flow/distance (-), slope (-), sand substrate (-), segment length (+), shade (-), and width to depth ratio (+). Percent shade was significant in both models with an inverse relationship with downstream temperature. Of these variables, percent shading was the only one linked to riparian vegetation type, and it had a negative relationship with downstream temperature. Since riparian trees providing more shade to streams than grasses, trees result in lower water temperatures compared to grasses. The percent shading variable has a coefficient of ~ -0.013 in both models (model 7 and 8; Table 7). This means if there is 100% shading over a stream segment the model would predict a 1.3°C decrease in stream temperatures as a result of riparian vegetation shading.

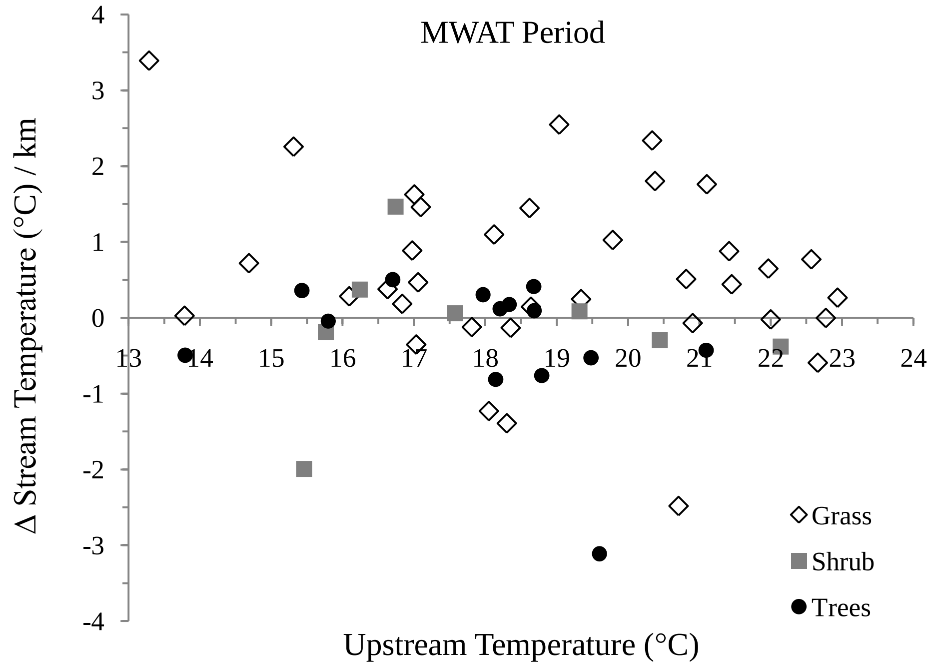
**Table 7.** Best multiple regression models for predicting downstream temperature during the MWAT time period in stream reaches from 12 trout streams located throughout north central Wisconsin. Model 8 explained the most variance and was selected as the best model, model 7 being an alternative model. See Appendix E for all models.

| Model | Maximum Weekly Average Temperature | Variable | | Intercept | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | (MWAT) Downstream temperature (°C) | Coefficient | P-value | Coefficient | P | Adj. $R^2$ | AIC | AICc | SBC |
| 7 | Upstream temperature (°C) | 0.76070 | <0.0001 | 4.78613 | <0.0001 | 0.916 | -106.78 | -105.06 | -83.60 |
| | Δ Flow/distance (cms/m) | -21358.00050 | <0.0001 | | | | | | |
| | Slope (m/m) | -324.33020 | <0.0001 | | | | | | |
| | Sand substrate (%) | -0.91031 | 0.0001 | | | | | | |
| | Segment length (m) | 0.00070 | 0.0002 | | | | | | |
| | Shade (%) | -0.01318 | <0.0001 | | | | | | |
| | LN Width (m) : depth (m) | 0.51486 | 0.0028 | | | | | | |
| 8 | Upstream temperature (°C) | 0.75122 | <0.0001 | 3.41483 | <0.0001 | 0.918 | -110.48 | -109.33 | -87.30 |
| | Δ Flow/distance (cms/m) | -26907.37202 | <0.0001 | | | | | | |
| | Slope (m/m) | -327.72774 | <0.0001 | | | | | | |
| | Shade (%) | -0.01281 | <0.0001 | | | | | | |
| | Segment length (m) | 0.00076 | <0.0001 | | | | | | |
| | LN Wetted width (m) | 0.91010 | <0.0001 | | | | | | |
| | LN Upstream flow (cms) | -0.40586 | <0.0001 | | | | | | |

Riparian Vegetation and Stream Temperature Responses

When corrected for upstream temperature during the MWAT period, no significant differences were present in the change in stream temperature per km of stream among the three riparian vegetation types (ANCOVA, F=3.052, P=0.055) (Figure 11a and 12a). However, during the MDAT period significant differences were present in the changes in stream temperature per km among the three riparian vegetation types (F=3.667, P=0.032). Despite differences in temperatures among vegetation types using the MDAT period temperatures, there was not a significant relation between the change in stream temperature per km of stream and the covariate, upstream temperature (F=3.438, P=0.069). As a result, when adjusted for upstream temperature in the analysis, the change in stream temperature per km is affected to a greater degree by the vegetation type as compared to the initial upstream temperature. Bonferroni's multiple comparison method found that grass riparian vegetation had a significantly higher change in stream temperature per km (stream temperatures increased faster) compared to tree vegetation (P=0.042), while shrub riparian vegetation type was not significantly different from either grass or tree vegetation (Figure 14a and 15a).

The change in stream temperatures measured per km of stream during the MWAT period ranged from -3.1 to 13.2°C/km with a mean value of 0.50°C/km (Figure 10). In contrast, the change in stream temperatures measured per km of stream during the MDAT period ranged from -4.6 to 15.2°C/km with a mean value of 0.64°C/km (Figure 13). The downstream temperature during the MWAT period ranged from 13.3 to 23.5°C and had a mean value of 18.8°C (Figure 10) and ranged from 13.3 to 25.4°C with a mean of 20.32°C during the MDAT period (Figure 13). Scatterplots of the data are presented showing the distribution of the data points for the



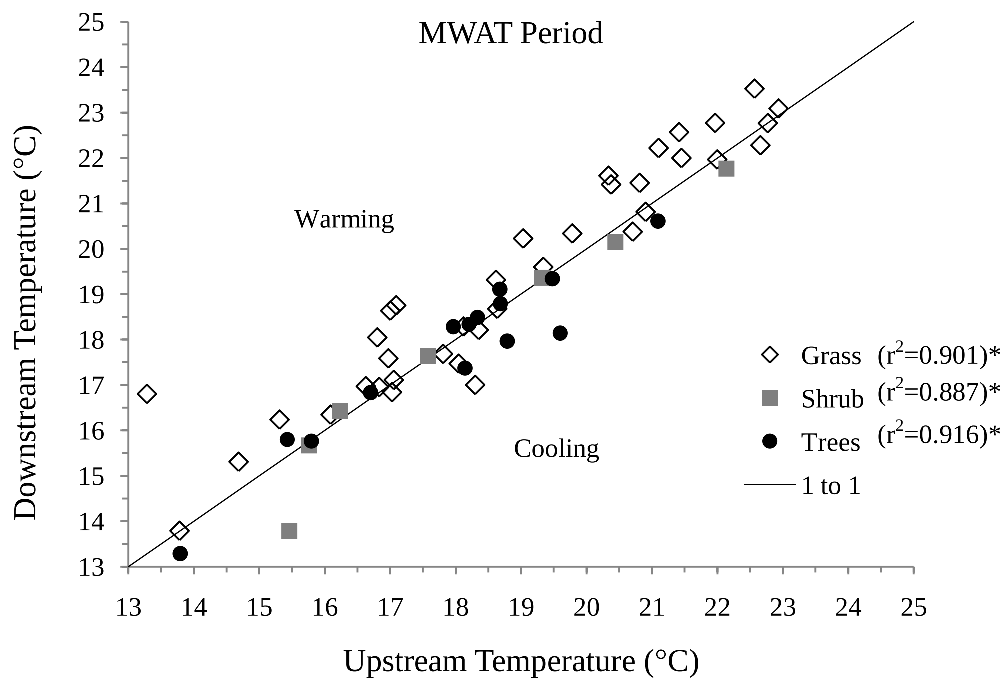
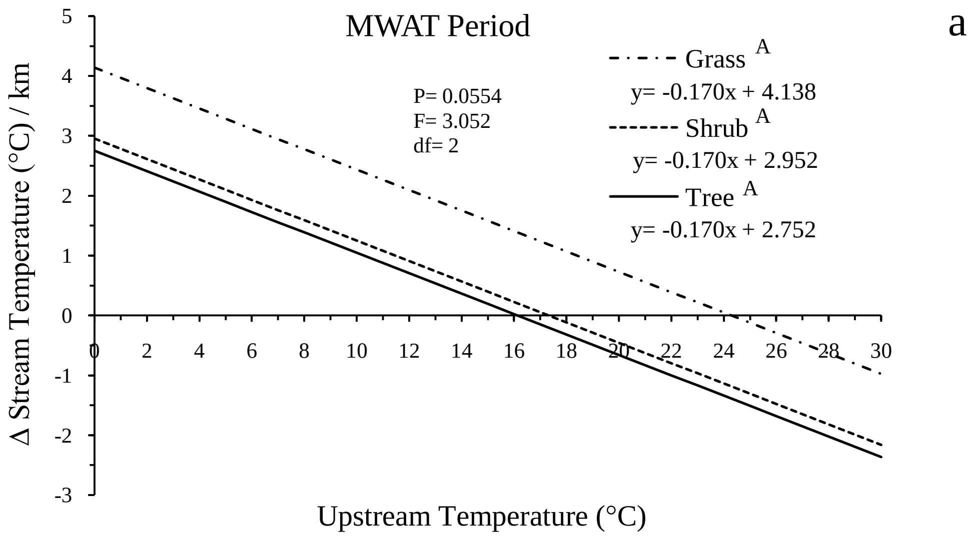




**Figure 10.** Scatterplots of actual change in stream temperatures per km and downstream temperatures measured as a function of initial upstream temperatures for each of the three riparian vegetation types at 59 stream segments (Grass n=37, Shrub n=8, and Tree n=14) across central Wisconsin during the MWAT period. Asterisk (*) indicate a significant correlation.



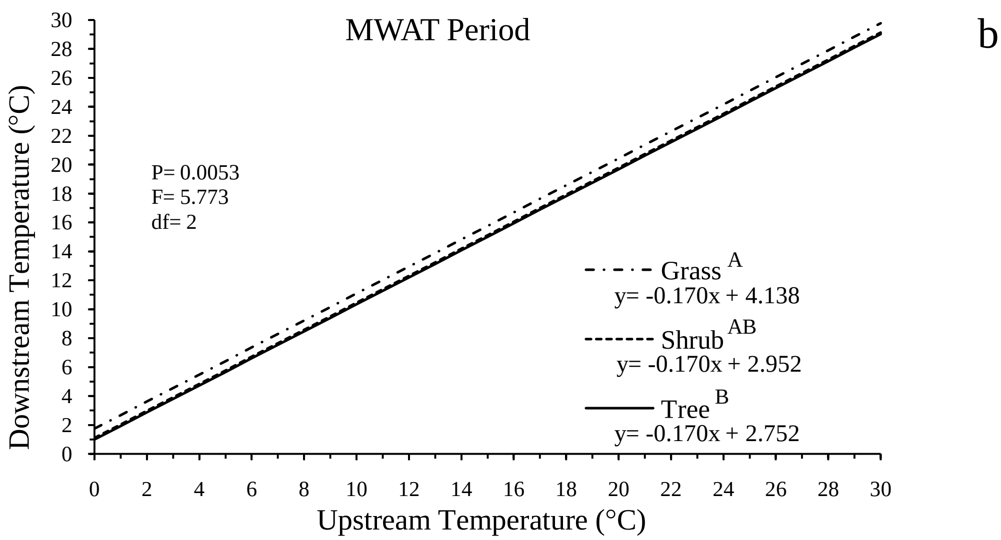
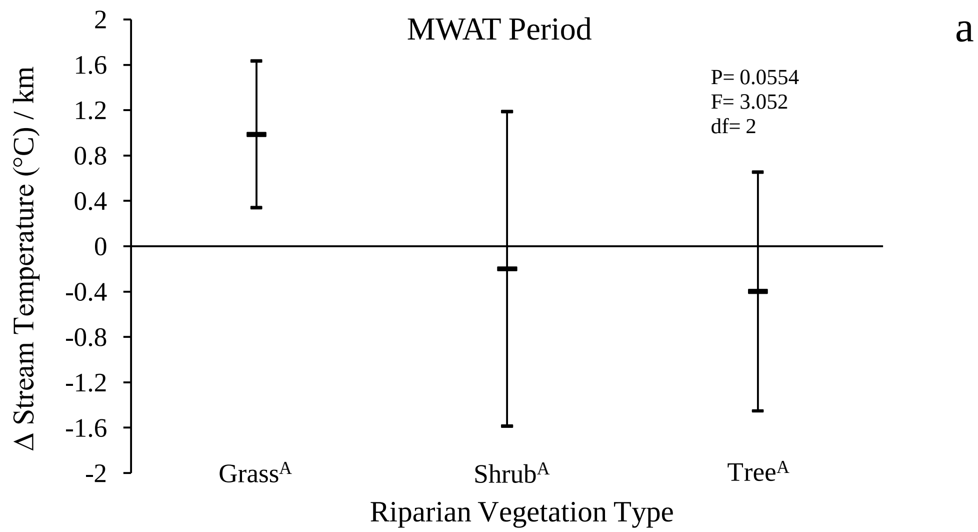




**Figure 11.** ANCOVA results for downstream temperature and change in stream temperature per km with upstream temperature as the covariate and vegetation type as the fixed factors from analyzing 59 stream segments (Grass n=37, Shrub n=8, and Tree n=14) across central Wisconsin during the MWAT period. Note the difference in scale for the dependent variables which affects the visual differences among vegetation types between the analyses.



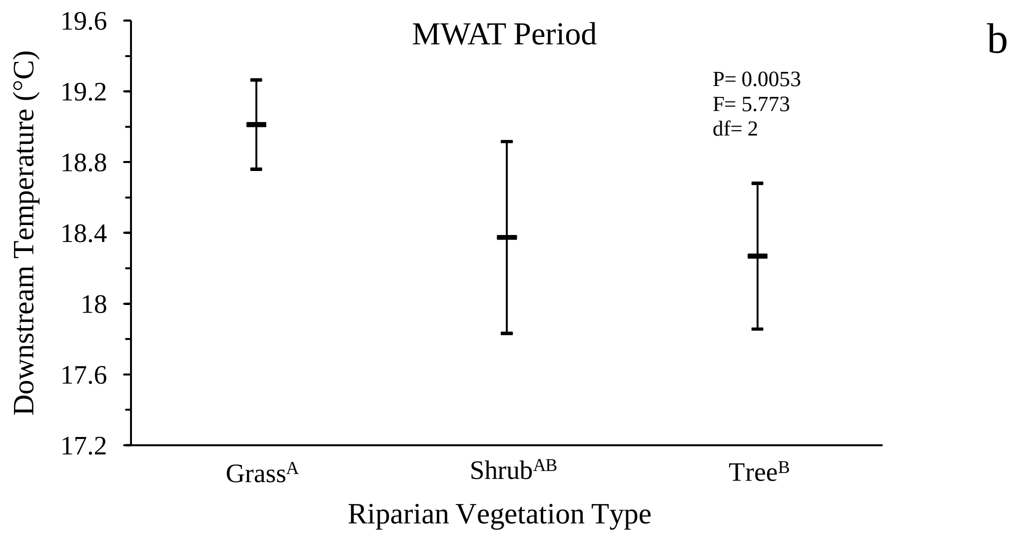




**Figure 12.** Estimated marginal means results of ANCOVA analyses for downstream temperature and change in stream temperature per km versus upstream temperature at 59 stream segments (Grass n=37, Shrub n=8, and Tree n=14) across central Wisconsin during the MWAT period. Model values represent the mean response evaluated with the covariate set at 18.5°C.

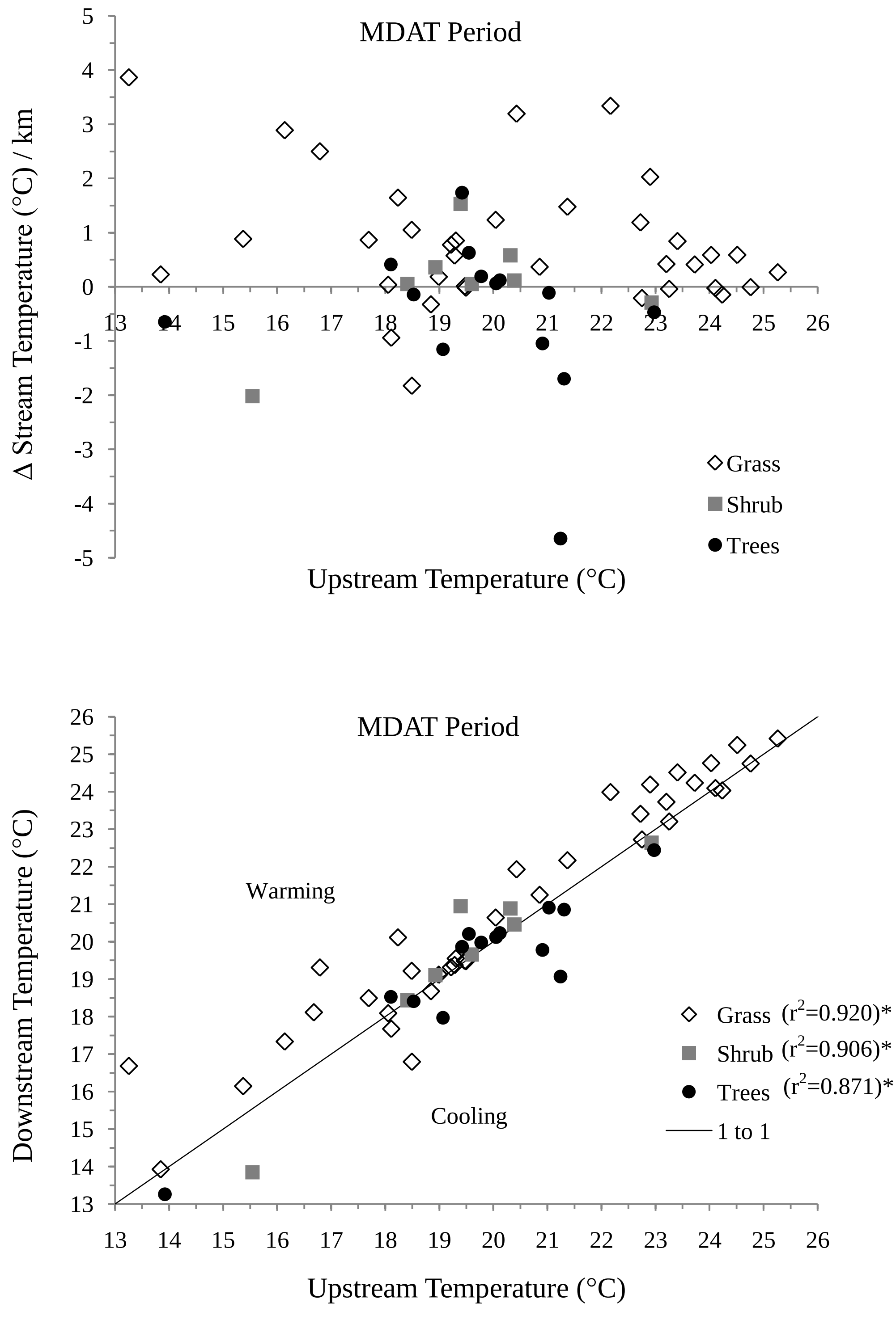
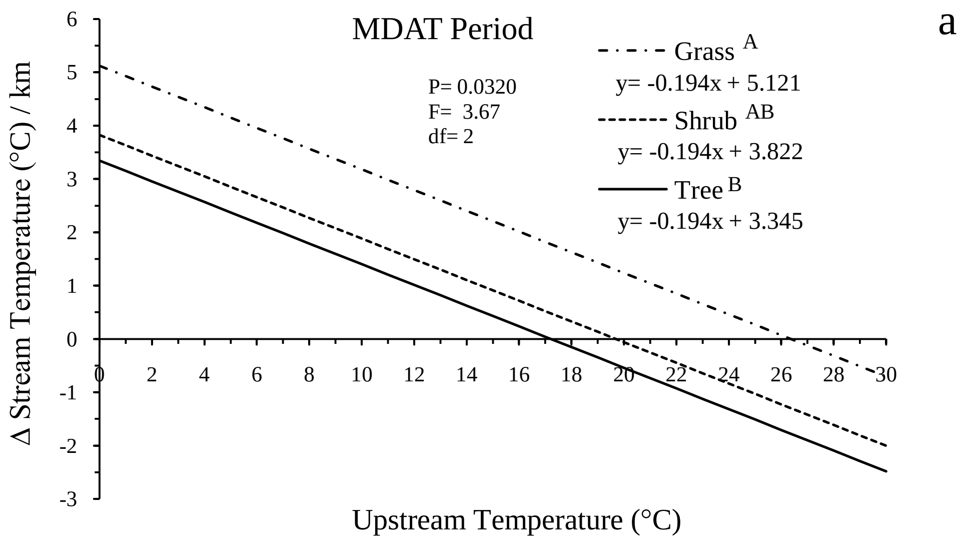



**Figure 13.** Scatterplots of actual change in stream temperatures per km and downstream temperatures measured as a function of initial upstream temperatures for each of the three riparian vegetation types at 59 stream segments (Grass n=37, Shrub n=8, and Tree n=14) across central Wisconsin during the MDAT period. Asterisk (*) indicate a significant correlation.



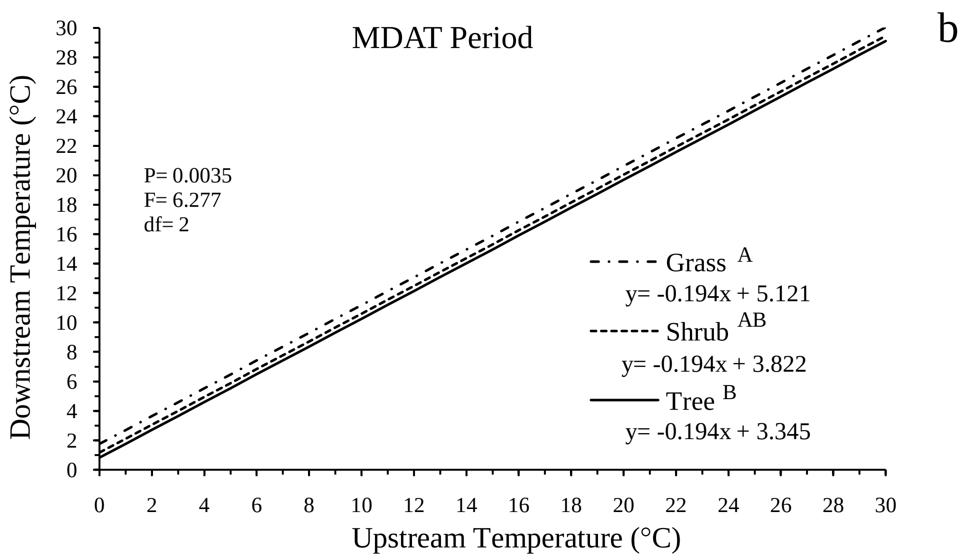




**Figure 14.** ANCOVA results for downstream temperature and change in stream temperature per km with upstream temperature as the covariate and vegetation type as the fixed factors from analyzing 59 stream segments (Grass n=37, Shrub n=8, and Tree n=14) across central Wisconsin during the MDAT period. Note the difference in scale for the dependent variables which affects the visual differences among vegetation types between the analyses.

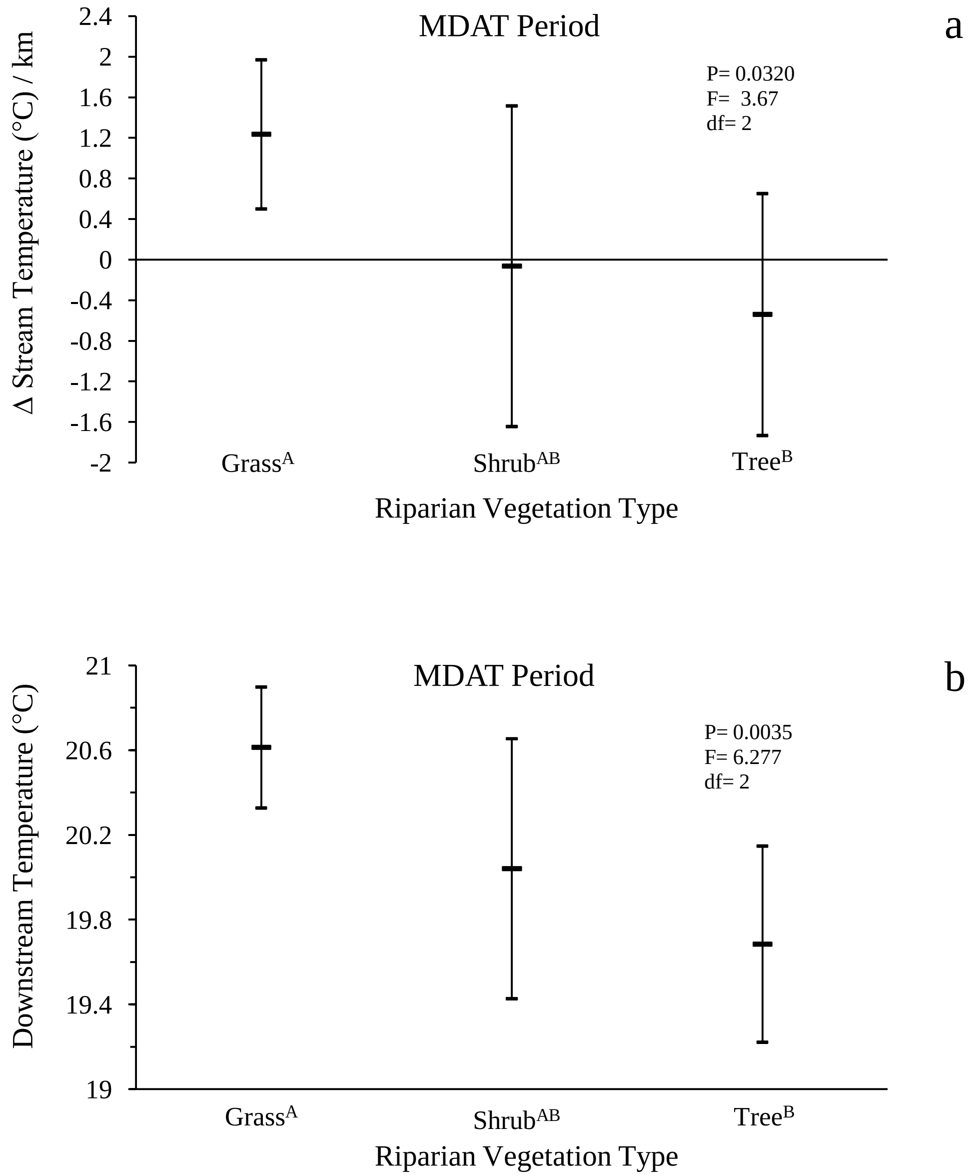



**Figure 15.** Estimated marginal means results of ANCOVA analyses for downstream temperature and change in stream temperature per km versus upstream temperature at 59 stream segments (Grass n=37, Shrub n=8, and Tree n=14) across central Wisconsin during the MDAT period. Model values represent the mean response evaluated with the covariate set at 20.0°C.

three riparian vegetation types in respect to the dependent variable and covariates (Figure 10 and 13).

An alternative analysis approach considered the final downstream temperature rather than the change in stream temperature per km of stream as the dependent variable. ANCOVA's with downstream temperature as the dependent variable corrected for upstream temperatures with riparian vegetation type as the fixed factors were run. During the MWAT period downstream temperatures were significantly different between vegetation types when accounting for upstream temperature ($F=5.773$, $P=0.005$) (Figure 11b and 12b), and the covariate, upstream temperature, was significantly related to downstream temperature ($F=496.105$, $P<0.0001$). After running Bonferroni's multiple comparisons between the vegetation types, tree vegetated segments had lower downstream temperatures compared to grass vegetated segments when accounting for upstream temperature ($P=0.010$). Shrub vegetated segments were not found to be significantly different from grass or tree segments. For the MDAT period there was also a significant difference between the vegetation types and downstream temperature with upstream temperature as the covariate ($F=6.277$, $P=0.004$) (Figure 14b and 15b), and there was a significant positive relation between the upstream temperature and the covariate, downstream temperature ($F=535.993$, $P<0.0001$). Comparing the main effects using Bonferroni's multiple comparison method, tree riparian vegetation had significantly lower temperatures than grass ($P=0.004$), but shrub vegetation was not significantly different from tree or grass riparian vegetation.

## DISCUSSION

This study assessed how riparian vegetation could affect stream temperatures in order to understand if changes in land use could increase the amount of thermally suitable habitat for brook trout. Because brook trout are strict thermal conformers and only survive across a narrow thermal range of colder water temperatures (MacCrimmon and Campbell 1969; Binns and Eiserman 1979; Elliot 1981; Bowlby and Roff 1986; Bozek and Hubert 1992; Stoneman and Jones 2000; Wehrly et al. 2003), modeling relations between riparian vegetation and stream temperature could be insightful for management agencies interested in protecting and/or increasing trout water. Weekly stream temperatures exceeding 22.3°C (Eaton et al. 1995), 23.3°C (Wehrly et al. 2007), and 24°C (Meisner 1990) and daily maximums in excess of 24°C (Binns and Eiserman 1979; Barton et al. 1985; Bowlby and Roff 1986; Picard et al. 2003) have been shown to limit trout distribution while brook trout prefer temperatures from 11.3 to 19°C (Cherry et al. 1975; Brungs and Jones 1977; Cherry et al. 1977).

In all study streams, a longitudinal warming trend was apparent as evidenced by their temporal thermal profiles; the exception was Walla Walla Creek. Walla Walla Creek was the only creek fed by a headwater lake resulting in warmer initial stream temperatures compared to the other study streams which were all spring-fed. Walla Walla Creek received large inputs of groundwater as it progressed downstream transitioning it to colder stream temperatures. This process of warm initial stream temperatures provided by headwater lakes followed by downstream cooling dominated by groundwater inputs has been identified in other areas as well (Mellina et al. 2002). Five of the twelve study streams passed the brook trout suitability threshold set at 22.3°C, where modeling was particularly insightful.

The SSTEMP model produced accurate depictions of the thermal trajectories for streams in central Wisconsin. Modeled stream temperatures during the MWAT period were within 0 to 1.2°C from observed temperatures. Temperature predictions were negatively impacted by the compounding factor of the model which used the predicted downstream temperature of one segment as the subsequent upstream inflow temperature in the next segment. This compounding factor leading to greater and greater deviances from actual measured stream temperatures in many cases. However, the model performed very well with only slight deviances from actual measured temperatures once calibrated. Other discrepancies could be the result of groundwater inflows which were adjusted in the model calibration. SSTEMP followed the actual measured stream temperatures with relatively minor calibrations. Unnamed 17-5 / Cunningham Creek was calibrated with maximum changes in modeled flow by as little as 0.0009 cms, which is well within the measurement errors made with the flow meter.

Similar heat budget stream temperature modeling has been able to accurately predict temperatures throughout the Upper Midwest region generally within 1°C (Sinokrot and Stefan 1993; Blann et al. 2002; Gaffield et al. 2005). For instance, Blann et al. (2002) calibrated a similar model, Stream Network Temperature Model (SNTEMP) (Theurer et al. 1984), and predicted stream temperatures that were just 0.12°C below actual measured temperatures at the validation points with a maximum error of -1.3°C across all temperature recordings over all time periods modeled. Gaffield et al. (2005) calibrated the SSTEMP model for southwestern Wisconsin streams and predicted stream temperatures within 0.5°C on Warner Branch and simulated temperatures deviated from measured temperatures by 1.0°C at most on Joos Creek.

The primary source of heat energy inputs into streams comes from direct solar radiation (Brown 1970; Beschta et al. 1987; Johnson 2004). Using heat budget analyses, Johnson (2004) found solar inputs on a non-shaded stream reach in the Oregon Cascade Range at 12:00 to account for 840 $W \cdot m^{-2}$ of the incoming energy fluxes compared to the next highest heat energy input of 18 $W \cdot m^{-2}$ from convection, with a net energy flux of 580 $W \cdot m^{-2}$ entering the stream. Conversely, under complete shading, the net energy flux was 149 $W \cdot m^{-2}$ with only 4 $W \cdot m^{-2}$ coming from solar inputs. Therefore, once the model was calibrated across both shaded and non-shaded riparian areas, I could simulate what the thermal profiles of these streams would look like if solar radiation heating was reduced through managing riparian vegetation for varying levels of shade. Shading levels of 0, 25, 50, and 75% were modeled to depict various degrees of shading that could thermally be managed for using different riparian vegetation types. Using this model, streams were predicted to lose as much as 91.54%, 5.7km, of their suitable trout water if shading was not present (0% shade). Conversely, streams which were not found to be completely suitable for brook trout gained as much as 55.26%, 3.21 km, if the streams were continuously shaded at the 75% level.

Equilibrium stream temperatures, the stream temperature achieved when the interaction of all heat fluxes through the water surface counterpoise each other to sum zero, decreased drastically as a result of increasing levels of shading. When 0% shade was modeled, the average temperature equilibrium of the eleven successfully modeled streams was 25.7°C, 25% shade resulted in an average equilibrium temperature of 23.9°C, 50% shade resulted in a temperature equilibrium of 22.6°C, and 20.9°C was the predicted equilibrium temperature under 75% stream shading. In southwest Wisconsin, Gaffield et al. (2005) predicted equilibrium temperatures of 29.1°C under 0% shade, 27.5°C under 30% shade, 26.4°C under 50% shade, 24.6°C under 80% shade, and 23.3°C when the stream was 100% shaded. Higher temperature equilibrium estimates by Gaffield et al. (2005) were likely due to warmer air temperatures and differences in the other meteorological variables used in their modeling. In general, Blann et al. (2002) estimated weekly average temperatures are 2.5°C higher for streams that pass through non-shaded reaches compared to shaded reaches using similar heat budget analysis. In order for riparian management to affect stream temperatures, other factors need to be considered. The length of stream necessary to reach equilibrium temperature for any given riparian condition is dependent upon stream size because the effects the high specific heat of water and water volume affect the rate of change in water temperature (Poole and Berman 2001). Rutherford et al. (2004) predicted that it takes 1,200 m for streams to reach equilibrium temperatures in 1 to 2 m wide streams in Australia and New Zealand and that streams with a width less than 2.5 m were provided the same amount of shade regardless of vegetation type. Moreover, it should be noted that once stream temperatures in open stream reaches exceed equilibrium temperatures in shaded reaches, shade can then act as a cooling effect to cool the stream. Otherwise shade allows the stream to warm at a slower rate. Decreasing stream temperatures can only occur as a result of groundwater inputs but shade plays a role in the magnitude of the cooling (Bartholow 2000).

Stream shading is not the only factor influencing stream temperatures based on different riparian vegetation types. Local microclimates have been shown to vary between riparian vegetation type and buffer width including air temperature, wind speed, and relative humidity (Barton et al. 1985; Bartholow 2000; Johnson 2004; Moore et al. 2005; Wilkerson et al. 2006). Maximum air temperatures have been found to be 3-4°C higher in unshaded areas directly adjacent to shaded areas (Johnson 2004). Moore et al. (2005) reviewed many papers documenting changes in microclimates between open and forested areas and found air temperatures at ground level to be as much as 10-15°C cooler in midday forested areas, wind speeds to be 10-20% of those in open areas, and midday humidity in forested areas to be 5-25%

higher than open areas. Wilkerson et al. (2006) assessed buffer width effects on stream temperature and found that buffer widths as small as 11 m showed minor but not significant increases in water temperature and buffer widths of 23 m showed no difference in water temperature compared to completely forested streams. Lengths of buffers affect stream temperature as well, with longer forested buffer strips leading to lower temperatures (Barton et al. 1985). Barton et al. (1985) determined that the portion of riparian area that was forested 2.5 km upstream of a site elucidated 56% of the variation in weekly maximum temperature. Differences in microclimates between tree and grass riparian-vegetated streams are simply not predictable enough to be accounted for appropriately in our modeling efforts. However, stream width and depth have been altered in temperature models in the Driftless Ecoregion of southwest Wisconsin where riparian grass vegetated streams have been found to be narrower and deeper (Trimble 1997; Stephens 2001; Gaffield et al. 2005). But in central Wisconsin riparian vegetation type has not been found to significantly affect stream width or depth and was therefore not altered in our modeling efforts (Stephens 2001; B. Cross, unpublished).

One stream, Gillespie Creek, was unable to be calibrated and thus accurately predict stream temperatures. The initial non-calibrated model created followed the pattern of the thermal profile very well yet it was on average 1.69°C warmer throughout its profile (Figure 7). There was no portion of this heat budget model that could explain the cooling effects that were observed. Hyporheic flow exchange between shallow groundwater and the water in the channel is likely the processes that would allow this stream to maintain colder than predicted temperatures. Hyporheic flow is not accounted for within the heat budget model because it is highly variable and relatively unpredictable. It has been theorized to cause cooling effects in streams resulting in colder than predicted stream temperatures in other studies (Johnsons and

Jones 2000; Poole and Berman 2001). Story et al. (2003) presented a similar case to Gillespie Creek where cooling stream reaches could not be accounted for by heat budget models without taking into account hyporheic exchange. Tracer dye tests and energy balance estimates were used to identify the exchange rate and temperature of hyporheic flows entering a 200 m long cooling stream reach. Their analysis concluded that a ~1.8°C cooling effect in stream temperatures was the result of hyporheic flow exchanges. One recent heat budget model was developed to include the flow exchanges between the channel and hyporheic zone, finding that the dominant cooling effect was hyporheic flow which provided 36 to 75% of the loss of heat energy in the streams (Rothwell et al. 2005).

Some previous research has encouraged removal of riparian trees and shrubs that shade stream channels in order to increase primary productivity in the streams leading to increased trout abundance and biomass (Hunt 1979; Peterson 1993). Hunt 1979 found that in central Wisconsin after removal of riparian brush (mostly speckled alder (*Alnus incana*)) trout abundance increased from 18% fewer trout per mile compared to the reference stream to 106% more wild brook trout per mile in the treated reach. However, it is important to note that temperature was clearly not a limiting factor in these streams and stream temperatures did increase by as much as 2.8°C throughout the 732 m long treatment reach likely due to the shade reduction caused by streamside woody vegetation removal. The increase in stream temperatures in these streams resulted in maximum temperatures that were still below the suitable threshold. In this study, Onemile, Spring / South Branch Pine, and Webster Creek might fall into this category of shade reduction leading to increased stream temperatures but not surpassing the brook trout thermal suitability threshold. It is evident that in all of the other study streams, any reduction in stream shade could lead to a decrease in summer trout distribution and decreases the length of water below 19°C which is the trout preferred threshold, as well as the length of water below the suitability threshold of 22.3°C. When removing shading vegetation, caution must be taken not to allow stream temperatures to exceed those preferred by species of coldwater fishes.

Multiple regression results demonstrated how close the correlation between upstream temperature and downstream temperature is and identified specific stream temperature components driving thermal dynamics of streams in central Wisconsin within the stream segments. Undoubtedly, upstream temperature was the most significantly related variable influencing downstream temperature. Change in discharge per length of stream was also significant because it represents evidence of infiltration of groundwater into the stream (most Wisconsin streams have gaining reaches or have no change in discharge). Addition of cold groundwater results in a cooling effect during the warmer summer months. Groundwater inputs were likely the main cooling effects evident in these study streams and have been identified as so in other studies (McRae and Edwards 1994; Stoneman and Jones 1996; Gaffield et al. 2005). Gaffield et al. (2005) modeled the distribution of groundwater inputs which emphasized the importance of groundwater on the thermal profile of a stream. Their modeling demonstrated that large inputs of groundwater occurring over short distances has a greater cooling effect compared to the same amount of groundwater being dispersed into the stream over a greater distance. But when groundwater inputs dissipated over greater distances, temperatures are moderated further downstream. In this study, slope was also a significant variable in predicting downstream temperatures and is also a variable considered in physics-based heat budget models (Thereur et al. 1984; Bartholow 1989). Donato (2002) found slope was significant in multiple linear regression modeling to predict the average recorded temperature of creeks in central Idaho. However, upon examination, it was found that streams with steeper slopes were more heavily shaded, possibly the result of trees outcompeting grasses in these reaches, and significant increases of topographic shading occurred when slopes increased.

In high landscape relief areas, topographic shading has been shown to play a significant role in the prediction of stream temperatures with increases in topographic shade resulting in lower predicted temperatures (Theurer et al. 1984; Bartholow 1989; Rutherford et al. 1997). Topographic shade plays a substantial role in moderating stream temperatures because it completely blocks out direct solar radiation when the sun's angle is less than the topographic angle (Rutherford et al. 1997). Shading from riparian vegetation has been documented to play a significant role in maintaining cooler stream temperatures (Beschta et al. 1987; Poole and Berman 2001; Moore et al. 2005; Cassie 2006; Webb et al. 2008). Two artificial stream shading experiments document the importance of shade for maintaining colder stream temperatures very nicely. Johnson et al. (2002) found that when shaded, the study stream located in the western Cascades Range of Oregon had maximum stream temperature 3-4°C lower compared to if it were not shaded. On Smith Creek in Virginia, Fink (2008) found that prior to artificial shading of a 550 m long stream reach the average daily mean stream temperature increased significantly by 0.2°C and after shading decreased significantly by 0.5°C. Therefore, it is no surprise that shading was identified as a cooling factor in this model as shading was inversely related to stream temperatures.

Other factors such as segment length, stream width, and upstream flow were found to influence downstream temperature predictions. Segment length was likely a significant variable in the model because stream temperature changes depend on how long water is exposed to specific environmental conditions and segments were reflecting homogenous reaches. However, in the model, its coefficient was not that large likely due to the relative lack of variability of segment lengths used to develop this model. The relationship is positive suggesting that the longer the stream length the warmer the downstream temperature and is logical because of the general warming trend apparent in the majority of our study streams. Stream width was also a significant variable in the model providing additional evidence that wider streams lead to warmer stream temperatures. Wider streams have larger surface areas exposed to heating effects such as convection with warmer air temperatures and solar radiation from the sun. Evaporation is also increased as streams widen but the heat energy lost through evaporation is minimal compared to the heat energy gained by convection and solar radiation (Brown 1969; Johnson 2004). The inverse relationship between upstream flow and downstream temperature in this model is likely due to the high specific heat of water not allowing larger streams to warm as quickly. With change in flow per km being significantly inversely related to downstream temperature increases, larger streams with higher amounts of groundwater inputs means larger decreases in stream temperatures (Poole and Berman 2001). Neumann et al. (2003) also found flow to be a significant variable that was inversely related to downstream temperature predictions, coinciding with our findings.

Interestingly, air temperature was not a significant variable in the empirical stream temperature model likely because it was not variable enough over the time period of these models and across the streams themselves. Variability was low because temperatures were all modeled for the MWAT time period which occurred over only one specific week. Had all the weekly average temperatures been modeled, air temperature would have been more variable and possible been a factor in the model. Increased correlation between air and water temperatures has been shown when data averaging time periods are increased from daily to monthly (Webb et al. 2008). The use of upstream temperature data also likely prodded the significance of air temperature as a variable in our model as it accounted for much of the effects air temperature has on water temperature. However, upstream temperature was included as a variable because without it the full understanding of the processes/variables most associated with downstream change in temperature would not be realized, because the rate of change in stream temperature is highly related to initial stream temperatures.

Overall multiple regression analysis explained 91.8% of the variation in changes in temperature from initial upstream to final downstream temperatures in the study streams, suggesting that this model could predict downstream temperatures very well and could be used as a reach scale stream temperature prediction model in central Wisconsin. However, it is important that this multiple regression model not be over-applied. Using the model in situations it was not designed for such as different time periods, larger rivers, steeper streams, or streams of different channel types is not recommended as these differences have been shown to greatly impact stream temperature changes (Evans et al. 1998).

Forested riparian areas had stream temperatures 0.74°C colder than grass-vegetated riparian areas at the downstream end of a segment when upstream temperature was accounted for during the MWAT period and 0.93°C colder during the MDAT period. In study streams, estimated marginal means for change in stream temperature per km were predicted to decrease 0.54°C/km under forested riparian areas compared to increasing 1.23°C/km under grass-vegetated riparian areas during the period of MDAT when upstream temperature was accounted for. During the period of MWAT, this relationship was not quite significant ($p = 0.055$) but the estimated marginal means for change in stream temperature per km were predicted to decline 0.40°C/km under forested riparian areas compared to increasing 0.99°C/km under grass-vegetated riparian areas. Stream temperatures were predicted to cool under riparian forests and warm under riparian grasses, indicating that if the riparian areas of these streams were completely forested, stream temperatures would not surpass suitable trout thresholds provided that initial suitable temperature stream temperatures were present and all other factors (e.g., discharge, stream size, average air temperatures) and stayed the same. Blann et al. (2002) found that in general, stream temperatures decreased along forested reaches and increased slightly in successional and grazed reaches suggesting that maximum temperatures were lower under forested conditions and that if all other conditions suitable for brook trout were met that they could be reintroduced.

## Management Recommendations

Efforts aimed at expanding brook trout distributions and increasing angling opportunities should focus on identifying streams thermally limiting to brook trout and managing them for more optimal thermal regimes to improve overall trout population health. Examination of temporal thermal profiles can be used to identify areas where stream temperatures are likely to limit the brook trout distributions. Using current large-scale stream temperature prediction models (i.e., Wehrly et al. 1998; Risley et al. 2003; Stewart et al.2006) to identify potential thermally degraded stream reaches would minimize the need for creating thermal profiles for all streams region-wide to those that could possibly benefit from reach-scale riparian restoration such shown by this study. Management agencies should identify target stream temperatures for brook trout, so when streams exceed these temperatures managing for colder thermal regimes would be a logical management objective.

This study used a rubric whereby trout distribution was limited by a MWAT of 22.3°C (Eaton et al. 1995). However, to appropriately manage for “healthy” trout populations, temperature goals should perhaps be set lower since brook trout tend to prefer temperatures

≤ 20°C (Cherry et al. 1975; Brungs and Jones 1977; Cherry et al. 1977; Wehrly et al. 1999). Setting a goal temperature of 22°C or less for the MDAT period should suffice for basic management (Lyons 1992; Lyons et al. 1996).

Stream temperature management should be considered an essential part of Wisconsin trout stream management. Creating a decision making tree to provide guidance on when and how to appropriately manage trout streams would be invaluable (Appendix J). Trout stream watersheds with increases in disturbance (agriculture, forestry, and urbanization) and/or trout streams lacking ample groundwater inputs to support a healthy trout population throughout the stream's entirety should be the focus of Wisconsin trout stream thermal management. Additionally, streams with satisfactory temperatures for trout should be managed in a way to maintain their temperatures. Factors influencing what thermal management actions should take place in these systems include stream size and land use activities. Smaller streams are influenced by riparian vegetation shading to a greater degree than larger streams. More recent land use activities that should be mitigated include urbanization and groundwater pumping for irrigation as they likely decrease groundwater inputs. Wetlands also provide challenges in managing stream temperatures. Shading a stream reach through a wetland can be impossible because riparian trees may not grow in these areas and therefore would not be a suitable management option. However, by providing adequate grass cover along stream corridors in wetlands and meadows coldwater species biological integrity can be improved compared to having cropland or grazed riparian areas (Marshall et al. 2008). Efforts to manage stream temperatures should consider the confounding influences these various land uses can have on stream temperatures.

Modeling increases of riparian vegetation shading predicted increases in the length of streams thermally suitable to brook trout by maintaining colder stream temperatures for longer stream distances. Clearly, forested riparian areas reduce stream temperatures more than riparian areas with grass vegetation (Hunt 1979; Barton et al. 1985; Blann et al. 2002; Moore et al. 2005; Fink 2008). Thus, managing for forested riparian vegetation alone can provide increases in suitable habitat for brook trout. Increased brook trout distribution could provide trout fishing opportunities in areas where, under grass riparian vegetation, there formerly was not a trout fishery. Colder stream temperatures can be attained in formerly thermally marginal grass areas by managing for trees which can lead to increases in trout abundance and population health. With more and better fishing opportunities being the goal of Wisconsin trout stream habitat management (WI DNR 2008), protecting riparian forests and managing for more of them should be primary management action if trout streams are thermally marginal or unsuitable; this is particularly true at the downstream reaches.

In addition to thermally enhancing streams for trout, many other long-term benefits to streams are achieved from riparian trees making restoration of mature riparian forests a logical management action in many situations. For example, providing natural continuous riparian vegetation buffers in non-forested watersheds has also been strongly linked to higher quality more desirable fish and macroinvertebrate populations (Stewart et al. 2001).

<u>Climate Change Implications</u>

Several studies have assessed the potential impacts climate change could have on stream temperatures (Meisner 1990; Sinokrot et al. 1995; Eaton and Sheller 1996; Poff et al. 1996; Pilgrim et al. 1998; Schindler 2001; Williams et al. 2009). Major implications to coldwater fisheries are expected as greenhouse gasses in the atmosphere increase resulting in increased air temperatures and greater variability in precipitation (Sinokrot and Stefan 1993). In addition to direct warming of surface water, increases in air temperature will cause increased groundwater temperatures leading to warmer initial stream temperatures (Meisner 1990) and directly increase stream temperature warming throughout the length of streams (Pilgrim et al. 1998). Also, increases in stream temperature could result indirectly as increased air temperatures alter riparian vegetation types (Sinokrot and Stefan 1993). In Wyoming Rahel et al. (1996) predicted 9-76% decreases in thermally suitable habitat for coldwater fishes with 1-5°C increases in air temperature. Eaton and Scheller (1996) predicted a 54.8% reduction in brook trout range throughout the United States based on projected summer air temperature increases. Possible population fragmentation within this range could also occur as trout are forced back into disparate headwater reaches that contain the only thermal refugia (Rahel et al. 1996).

This study addressed needs relevant to climate change including the identification of thermal drivers in stream fish community zonation and the shifts that might occur due to land use changes and various levels of stream shading. However, by specifically targeting the MWAT period, possibly fewer influences of the effects of increased air temperatures on fish community zonation were identified than exist. Still, this study did recognize the critical need for cold groundwater temperatures and adequate stream flows, which have the potential to be altered in predicted climate change scenarios (IPPC 2007), as well as the importance of shade in providing cooler stream temperatures. By identifying stream reaches where potential increases in stream temperatures are most likely to occur, management efforts can be focused on these reaches to decrease potential negative impacts on coldwater species associated with elevated stream temperatures.

Models to identify areas of streams most subject to warming and assessments of the impacts of riparian vegetation on stream temperatures were used to address some concerns of the Wisconsin Initiative on Climate Change Impacts Coldwater Fisheries Working Group who were charged with investigating management adaptation strategies in response to climate change impacts (M. Mitro, Wisconsin Department of Natural Resources, personal communication). Coldwater stream reaches in Wisconsin most subject to climate change are those with the least amount of groundwater input relative to the amount of flow in the stream. Increases in air temperatures would cause increases in groundwater temperatures, however if groundwater temperatures increase only slightly they would still provide stream temperatures appropriate for coldwater fishes with relatively small losses in the length of stream suitable to these species. Predictions in the changes in precipitation are not as well defined yet, but these changes could also influence stream temperatures. Less precipitation would result in less groundwater recharge and in turn less groundwater inputs to streams. Also, if precipitation does come in large quantities but is restricted to shorter periods (such as the fall of 2007 and spring of 2008), groundwater recharge may be negatively impacted due to the fact that most precipitation becomes runoff and does not infiltrate into the groundwater table. Increased flow events could result in larger wetted widths as well, possibly leading to increased convection with warmer air temperatures and increased amounts of solar radiation received by a stream.

Management for progressive riparian vegetation succession into older growth riparian forests in central Wisconsin may be one management strategy for maintaining cooler stream temperatures in the face of climate change. This project identified that trees in the riparian areas provide and maintain colder stream temperatures compared to grasses. Other management strategies leading to a decrease in the surface area of a stream exposed to solar radiation may also decrease stream temperatures or maintain them if and/or when increased air temperatures occur due to climate change.

Potential impacts of climate change on coldwater fisheries in Wisconsin should be assessed by modeling different scenarios of climate change on stream temperatures (i.e., changes in air temperatures, groundwater temperatures, precipitation, etc…). Williams et al. (2009) undertook such a task for the Western U.S. and identified areas of concern for three species of cutthroat trout (*Oncorhynchus clarkia*) due to potential climate change impacts. Fisheries management adaptation strategies addressing climate change in Wisconsin could be focused on stream reaches where such models show coldwater fish species persistence is at risk.

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**APPENDICES**

**Appendix A.** SSTEMP calibrations.

| Segment # | Δ Inflow (cfs) | Δ Outflow (cfs) | Δ Accretion Temperature (°C) | Δ Width's A term | Δ Total Shade (%) |
|---|---|---|---|---|---|
| **Unnamed 17-5 / Cunningham Creek** | | | | | |
| 1 and 2 | Not Modeled | Not Modeled | Not Modeled | Not Modeled | Not Modeled |
| 3 | 0.0124 | 0.0145 | 0 | 0 | 0 |
| 4 and 5 | Combined | Combined | Combined | Combined | Combined |
| 6 and 7 | Combined | Combined | Combined | Combined | Combined |
| 8 | 0 | 0 | 0 | 0 | 0 |
| 9 | 0 | 0 | 0 | 0 | 0 |
| 10 and 11 | | | | | |
| Trib input | -0.0230 | 0 | 0 | 0 | 0 |
| 12 | 0 | -0.0317 | 2.5 | 0 | 0 |
| 13 | -0.0317 | 0 | 0 | 0 | 0 |
| **Blake Creek** | | | | | |
| 1 | 0 | 0.2407 | 0 | 0 | 0 |
| 2 | 0.2377 | 0.3527 | 0 | 0 | 0 |
| 3 | 0.3527 | 0 | 3.055 | 0 | 0 |
| 4 | 0 | -0.2610 | 3.055 | 0 | 0 |
| Trib input | -0.2610 | 0 | 0 | 0 | -18.51 |
| 5 | -4.9135 | -1.6890 | 0 | 0 | -18.51 |
| 6 | -1.6890 | -0.6771 | 0 | 0 | 0 |
| 7 | -0.6771 | 0 | 0 | 0 | 0 |
| Trib input | 0 | 0 | 0 | 0 | 0 |
| 8 to 10 | Combined | Combined | Combined | Combined | Combined |
| 11 | 0 | -0.0891 | 0 | 0 | 0 |
| 12 | -0.0891 | 0.1037 | 0 | 0 | 0 |
| 13 | 0.1037 | 0.1000 | 0 | 0 | 0 |
| 14 | 0.1000 | 0.1430 | 0 | 0 | 0 |
| 15 | 0.1430 | -0.2782 | 0 | 0 | 0 |
| 16 | -0.2782 | -0.0600 | 0 | 0 | 0 |
| 17 | -0.0600 | 0 | 0 | 0 | 0 |
| 18 | 0 | 0 | 0 | 0 | 0 |
| 19 | 0 | 0 | 0 | 0 | 0 |
| **Magdanz / Hatton Creek** | | | | | |
| 1 | 0 | 0.1525 | 0 | 0 | 0 |
| 2 | 0.1525 | 0 | 0 | 0 | 0 |
| 3 | 0 | 0.0108 | 0 | 0 | 0 |
| 4 | 0.0108 | 0 | 0 | 0 | 0 |
| 5 | 0 | 0 | 0 | 0 | 0 |
| 6 | 0 | 0.1104 | 0 | 0 | 0 |
| 7 | 0.1104 | 0.1759 | 0 | 0 | 0 |

**Appendix A.** (Continued)

| Segment # | Δ Inflow (cfs) | Δ Outflow (cfs) | Δ Accretion Temperature (°C) | Δ Width's A term | Δ Total Shade (%) |
|---|---|---|---|---|---|
| **Magdanz / Hatton Creek (Continued)** | | | | | |
| 8 | 0.1759 | 0.2210 | 0 | 0 | 0 |
| 9 | 0.2210 | -0.0787 | 0 | 0 | 0 |
| 10 | -0.0787 | 0 | 0 | 0 | 0 |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 11 | 0 | 0 | 3.0550 | 0 | 0 |
| 12 | 0 | 0 | 7.0550 | 0 | 0 |
| **Little Wolf River (North Branch)** | | | | | |
| 1 | 0 | 0 | 0 | 0 | 0 |
| 2 | 0 | 0 | 0 | 0 | 0 |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 3 | 0 | 0 | 0 | 0 | 0 |
| 4 | 0 | -0.0599 | 0 | 0 | 0 |
| 5 | -0.0599 | -0.0975 | 0 | 0 | 0 |
| 6 | -0.0975 | -0.3180 | 0 | 0 | 0 |
| 7 | -0.3180 | -0.4194 | 0 | 0 | 0 |
| 8 | -0.4194 | -0.3544 | 0 | 0 | 0 |
| 9 | -0.3544 | -0.3468 | 0 | 0 | 0 |
| 10 | -0.3468 | -0.1302 | 0 | 0 | 0 |
| 11 | -0.1302 | -0.1422 | 0 | 0 | 0 |
| 12 | -0.1422 | -0.1717 | 0 | 0 | 0 |
| 13 | -0.1717 | -0.2017 | 3.0556 | 0 | 0 |
| 14 | -0.2017 | -0.1626 | 3.0556 | 0 | 0 |
| 15 | -0.1626 | 0.0440 | 3.0556 | 0 | 0 |
| 16 | 0.0440 | 0.1275 | 3.0556 | 0 | 0 |
| 17 | 0.1275 | 0.0762 | 3.0556 | 0 | 0 |
| **Sucker Creek** | | | | | |
| 1 | 0 | 0.2570 | 0 | 0 | 0 |
| 2 | 0.2570 | 1.2318 | 0 | 0 | 0 |
| Trib Input | 1.2318 | 0 | 0 | 0 | 0 |
| 3 | 0 | 0 | 0 | 0 | 0 |
| 4 | 0 | 0.0447 | 0 | 0 | 0 |
| 5 | 0.0447 | 0.3627 | 0 | 0 | 0 |
| 6 | 0.3627 | 0.4373 | 0 | 0 | 0 |
| 7 | 0.4373 | 0.4747 | 0 | 0 | 0 |
| 8 | 0.4747 | 0.5091 | 0 | 0 | 0 |
| 9 | 0.5091 | 0.4670 | 0 | 0 | 0 |
| 10 | 0.4670 | 0 | 0 | 0 | 0 |
| 11 | 0 | -0.0231 | 0 | 0 | 0 |

**Appendix A.** (Continued)

| Segment # | Δ Inflow (cfs) | Δ Outflow (cfs) | Δ Accretion Temperature (°C) | Δ Width's A term | Δ Total Shade (%) |
|---|---|---|---|---|---|
| **Sucker Creek (Continued)** | | | | | |
| 12 | -0.0231 | 0.0124 | 0 | 0 | 0 |
| 13 | 0.0124 | 0.0147 | 0 | 0 | 0 |
| 14 | 0.0147 | 0.0317 | 0 | 0 | 0 |
| 15 | 0.0317 | 0.0333 | 0 | 0 | 0 |
| 16 | 0.0333 | -0.0180 | 0 | 0 | 0 |
| 17 | -0.0180 | 0 | 0 | 0 | 0 |
| **Shioc River (West Branch)** | | | | | |
| 1 | 0 | 0.0706 | 0 | 0 | 0 |
| 2 | 0.0706 | 0.0779 | 0 | 0 | 0 |
| 3 | 0.0779 | 0.1249 | 0 | 0 | 0 |
| 4 | 0.1249 | -0.1431 | 0 | 0 | 0 |
| 5 | -0.1431 | 0 | 0 | 0 | 0 |
| **Shioc River (West Branch) (Continued)** | | | | | |
| 6 | 0 | 0 | 0 | 0 | 0 |
| 7 | 0.1106 | 0.0050 | 0 | 0 | 0 |
| 8 | 0.0050 | 0.0018 | 0 | 0 | 0 |
| 9 | 0.0018 | 0.0249 | 0 | 0 | 0 |
| 10 | 0.0249 | 0.0938 | 0 | 0 | 0 |
| 11 | 0.0938 | 0.0951 | 0 | 0 | 0 |
| 12 | 0.0951 | 0.0330 | 0 | 0 | -28.19 |
| 13 | 0.0330 | -0.0151 | 0 | 0 | -49.00 |
| 14 | -0.0151 | -0.0355 | 0 | 0 | 0 |
| 15 | -0.0355 | -0.0383 | 0 | 0 | 0 |
| **Bronken Creek** | | | | | |
| 1 | -0.0214 | 0 | 0 | 0 | 0 |
| 2 | 0 | 0.1255 | 0 | 0 | 0 |
| 3 | 0.1255 | 0.7502 | 0 | 0 | 0 |
| 4 | 0.7502 | 0.0530 | 0 | 0 | 31.25 |
| 5 | -0.6970 | -0.7103 | 0 | 0 | 0 |
| 6 | -0.1603 | 0 | 0 | 0 | 0 |
| 7 | 0 | 0 | 0 | 0 | 0 |
| 8 | 0 | 0.3903 | 0 | 0 | 0 |
| 9 | 0.3903 | 0.5066 | 0 | 0 | 0 |
| 10 | 0.5066 | 0.5982 | 0 | 0 | 0 |
| 11 | 0.5982 | -0.1312 | 0 | 0 | 0 |
| 12 | -0.1312 | 0 | 0 | 0 | 0 |
| **Onemile Creek** | | | | | |
| 1 | 0 | 0.4017 | 0 | 0 | 0 |

**Appendix A.** (Continued)

| Segment # | Δ Inflow (cfs) | Δ Outflow (cfs) | Δ Accretion Temperature (°C) | Δ Width's A term | Δ Total Shade (%) |
|---|---|---|---|---|---|
| **Onemile Creek (Continued)** | | | | | |
| 2 | 0.4017 | 0.4484 | 0 | 0 | 0 |
| 3 | 0.4484 | 0.6808 | 0 | 0 | 0 |
| 4 | 0.6808 | -0.1692 | 0 | 0 | 0 |
| 5 | -0.1692 | -0.1469 | 0 | 0 | 0 |
| 6 | -0.1469 | 0.6436 | 0 | 0 | 0 |
| 7 | 0.6436 | 0 | 0 | 0 | 0 |
| 8 | 0 | -0.3344 | 0 | 0 | 0 |
| 9 | -0.3344 | -1.2412 | 0 | 0 | 0 |
| 10 | -1.2412 | -0.9000 | 0 | 0 | 0 |
| 11 | -0.9000 | -1.3417 | 0 | 0 | 0 |
| 12 | -1.3417 | -0.6301 | 0 | 0 | 0 |
| 13 | -0.6301 | 0.1515 | 0 | 0 | 0 |
| 14 | 0.1515 | 0 | 0 | 0 | 0 |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 15 | 0 | -3.0400 | 0 | 0 | 0 |
| 16 | -3.0150 | -1.7425 | 0 | 0 | 0 |
| 17 | -1.7425 | 0 | 0 | 0 | 0 |
| 18 | 0 | -0.4000 | 0 | 0 | 0 |
| 19 | -0.4000 | -0.8491 | 0 | 0 | 0 |
| **Spring / South Pine Creek** | | | | | |
| 1 | 0 | 0 | 0 | 0 | 12.14 |
| 2 | 0 | 0 | 0 | 0 | 43.90 |
| 3 | 0 | 0 | 0 | 0 | 38.10 |
| 4 | 0 | 0.0090 | 0 | 0 | 62.50 |
| 5 | 0.0090 | 0.0007 | 0 | 0 | 53.10 |
| Trib Input | 0.0007 | 0 | 0 | 0 | 0 |
| 6 | 0 | 0 | 0 | -6.5425 | 0 |
| 7 | 0 | -0.2437 | 4.9444 | 0 | 0 |
| 8 | -0.2437 | -0.1083 | 4.9444 | 0 | 0 |
| 9 | -0.1083 | 0 | 1.7444 | 0 | 0 |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 10 | 0 | 0 | 4.0744 | 0 | 0 |
| 11 | 0 | -0.1539 | 0.4444 | 0 | 0 |
| 12 | -0.1539 | -0.3290 | 0.9444 | 0 | 0 |
| 13 | -0.3290 | -0.6789 | 0.9444 | 0 | 0 |
| 14 | -0.6789 | 0 | 0 | 0 | 0 |
| 15 | 0 | -0.4953 | 5.4444 | 0 | 0 |
| 16 | -0.4953 | -0.2414 | 5.4444 | 0 | 0 |

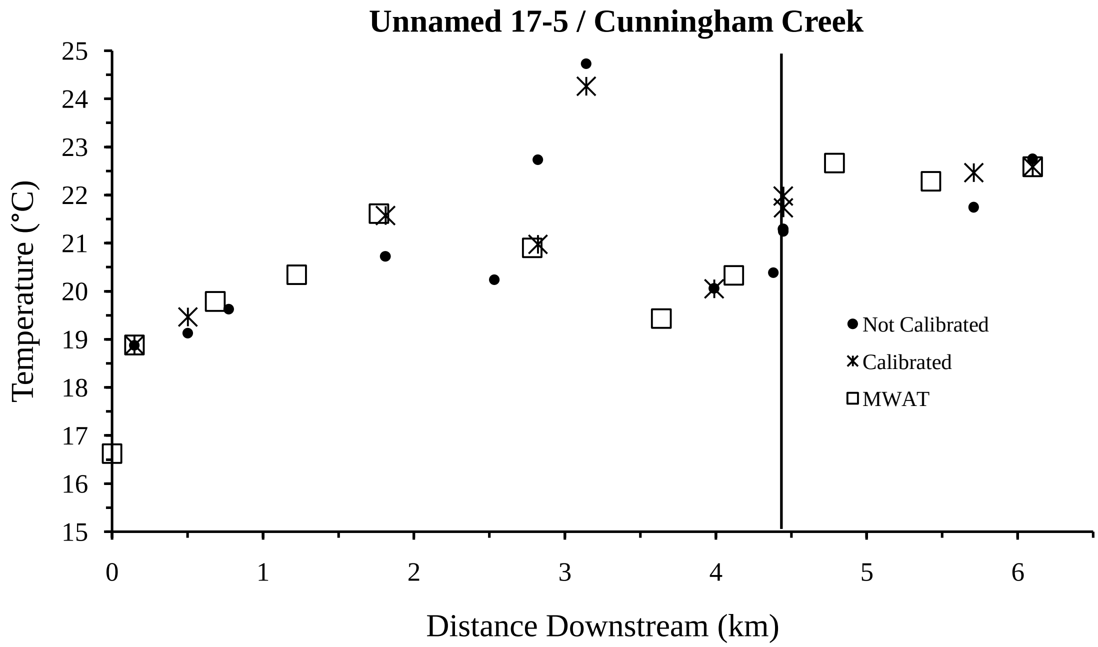
**Appendix A.** (Continued)

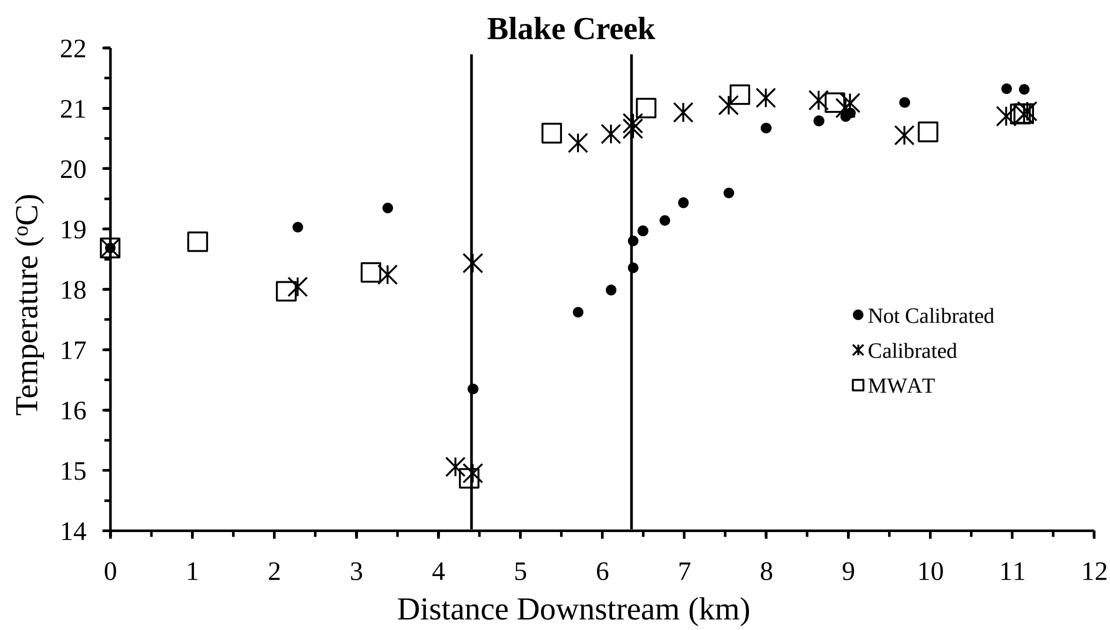
| Segment # | Δ Inflow (cfs) | Δ Outflow (cfs) | Δ Accretion Temperature (°C) | Δ Width's A term | Δ Total Shade (%) |
|---|---|---|---|---|---|
| **Spring / South Pine Creek (Continued)** | | | | | |
| 17 | -0.2414 | 0 | 0 | 3.1944 | 0 |
| **Walla Walla Creek** | | | | | |
| 1 | 0.5349 | 0.2682 | 0 | 0 | 0 |
| 2 | 0.2682 | 0 | 0 | 0 | 0 |
| 3 | 0 | -0.1761 | 4.0556 | 0 | 0 |
| 4 | -0.1741 | -0.1343 | 0 | 0 | 0 |
| 5 | -0.1343 | -0.1243 | 0 | 0 | 0 |
| 6 | -0.1243 | -0.1718 | 4.0556 | 0 | 0 |
| 7 | -0.1718 | -0.0642 | 0 | 0 | 0 |
| 8 | -0.0642 | -0.3834 | 0 | 0 | 0 |
| 9 | -0.3834 | 0.0924 | 0 | 0 | 0 |
| 10 | 0.0924 | -0.3915 | 4.0556 | 0 | 0 |
| 11 | -0.3915 | -0.2725 | 5.5556 | 0 | 0 |
| Trib Input | -0.2725 | -0.2725 | 0 | 0 | 0 |
| 12 | -0.2725 | -0.5129 | 4.0556 | 0 | 0 |
| 13 | -0.5129 | -0.2708 | 4.0556 | 0 | 0 |
| 14 | -0.2708 | 0 | 4.0556 | 0 | 0 |
| 15 | 0 | -0.2260 | 4.0556 | 0 | 0 |
| 16 | -0.2260 | -0.2408 | 4.0556 | 0 | 0 |
| 17 | -0.2408 | 0.0232 | 0 | 0 | 0 |
| 18 | 0.0232 | 0.3941 | 0 | 0 | 0 |
| 19 | 0.3941 | 0 | 0 | 0 | 0 |
| 20 | 0 | 0.1871 | 4.0556 | 0 | 0 |
| 21 | 0.1871 | -0.0582 | 4.0556 | 0 | -10.80 |
| 22 | -0.0582 | -0.2249 | 4.0556 | 0 | -9.40 |
| 23 | -0.2249 | -0.1832 | 4.0556 | 0 | -7.50 |
| **Webster Creek** | | | | | |
| 1 | 0 | -0.0465 | 2.8056 | 0 | 0 |
| 2 | -0.0465 | 0.1494 | 2.8056 | 0 | 0 |
| 3 | 0.1494 | 0.0964 | 2.8056 | 0 | 0 |
| 4 | 0.0964 | 0 | 2.8056 | 0 | 0 |
| 5 | 0 | -0.1032 | 0 | 0 | 0 |
| 6 | -0.1032 | 0 | 0 | 0 | 0 |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 7 | 0 | -0.0203 | 2.8056 | 0 | 0 |
| 8 | -0.0203 | -0.0338 | 0 | 0 | 0 |
| 9 | -0.0338 | -0.1084 | 2.8056 | 0 | 0 |
| 10 | -0.1084 | 0 | 0 | 0 | 0 |

**Appendix A.** (Continued)

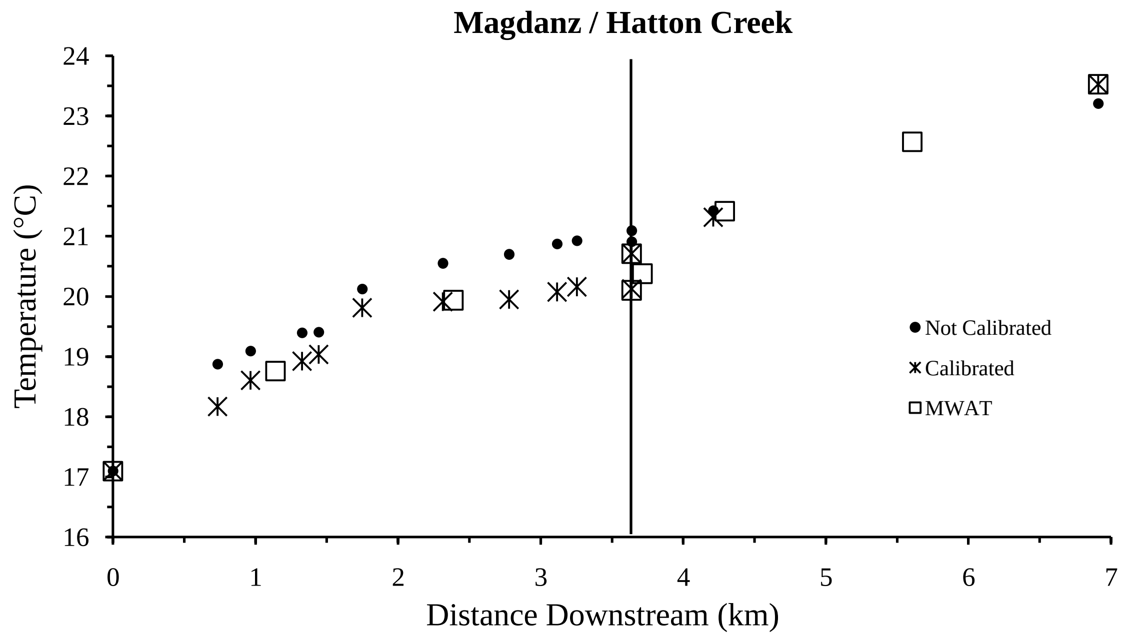
| Segment # | Δ Inflow (cfs) | Δ Outflow (cfs) | Δ Accretion Temperature (°C) | Δ Width's A term | Δ Total Shade (%) |
|---|---|---|---|---|---|
| **Webster Creek (Continued)** | | | | | |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 11 | 0 | -0.0344 | 0 | 0 | 0 |
| 12 | -0.0344 | 0 | 0 | 0 | 0 |
| 13 | 0 | 0.2994 | 0 | 0 | 0 |
| 14 | 0.2994 | 0.2334 | 0 | 0 | 0 |
| Trib Input | 0 | 0 | 0 | 0 | 0 |
| 15 | 0.2334 | 0 | 2.8056 | 0 | 0 |
| 16 | 0 | 0 | 2.8056 | 0 | 0 |
| 17 | 0 | 0.2706 | 2.8056 | 0 | 0 |
| 18 | 0.2706 | 0 | 2.8056 | 0 | 0 |

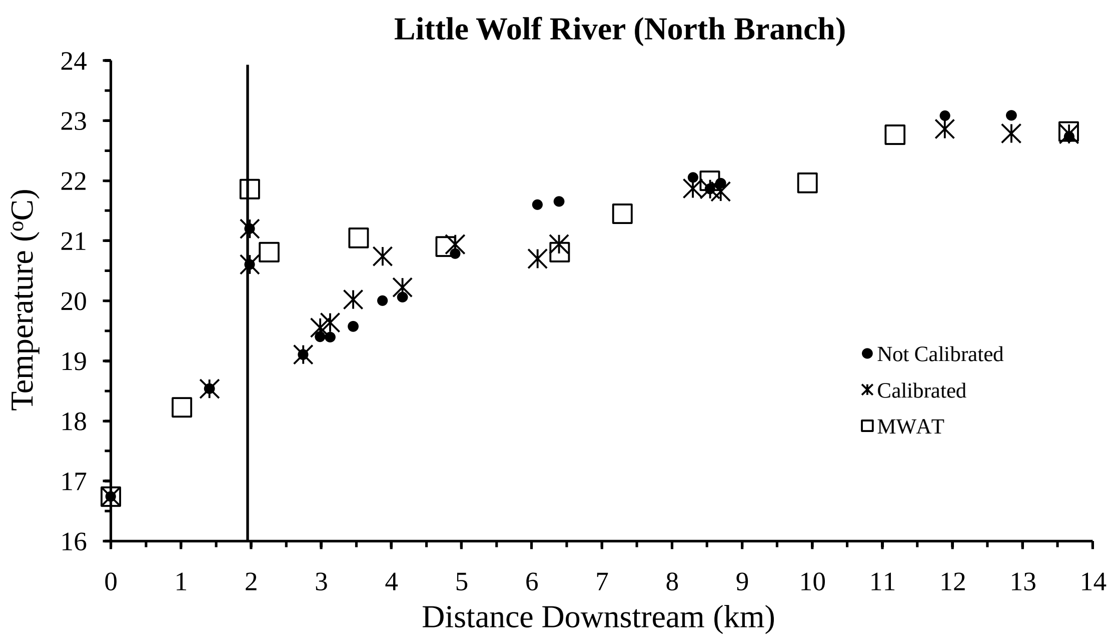
**Appendix B.** MWAT compared to calibrated and non calibrated SSTEMP model predictions. Vertical lines represent locations of tributaries entering the streams.



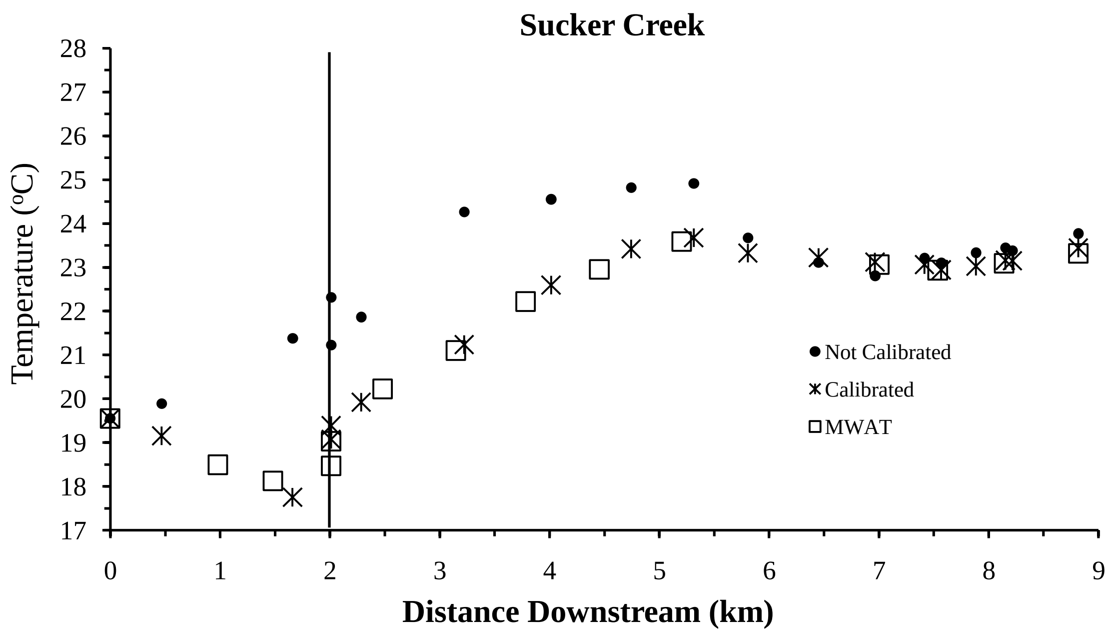



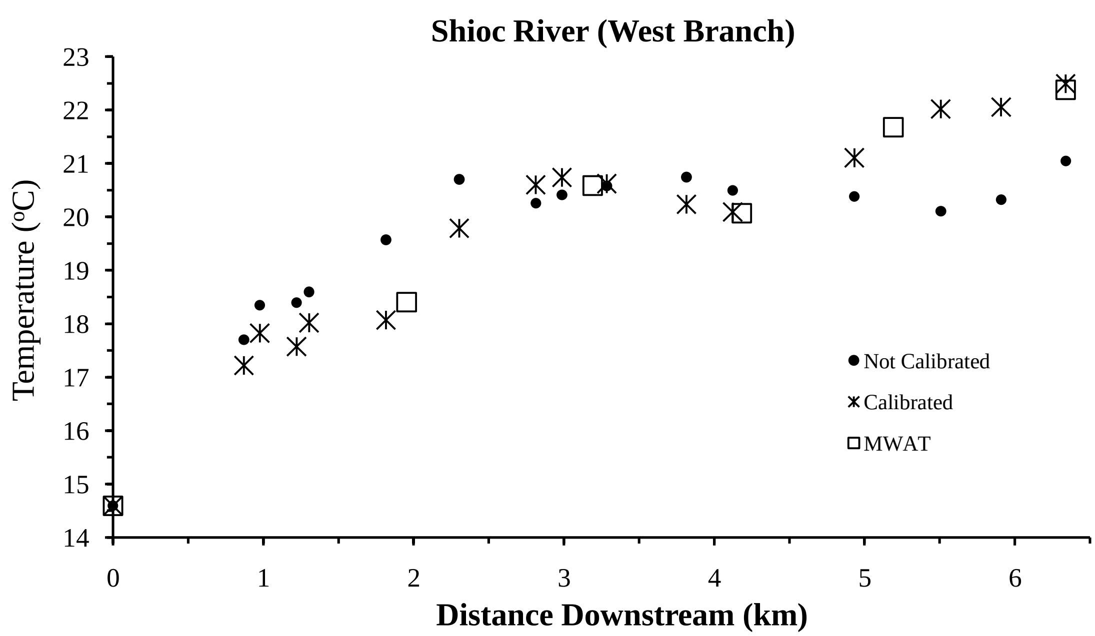
**Appendix B.** (Continued) MWAT compared to calibrated and non calibrated SSTEMP model predictions. Vertical lines represent locations of tributaries entering the streams.



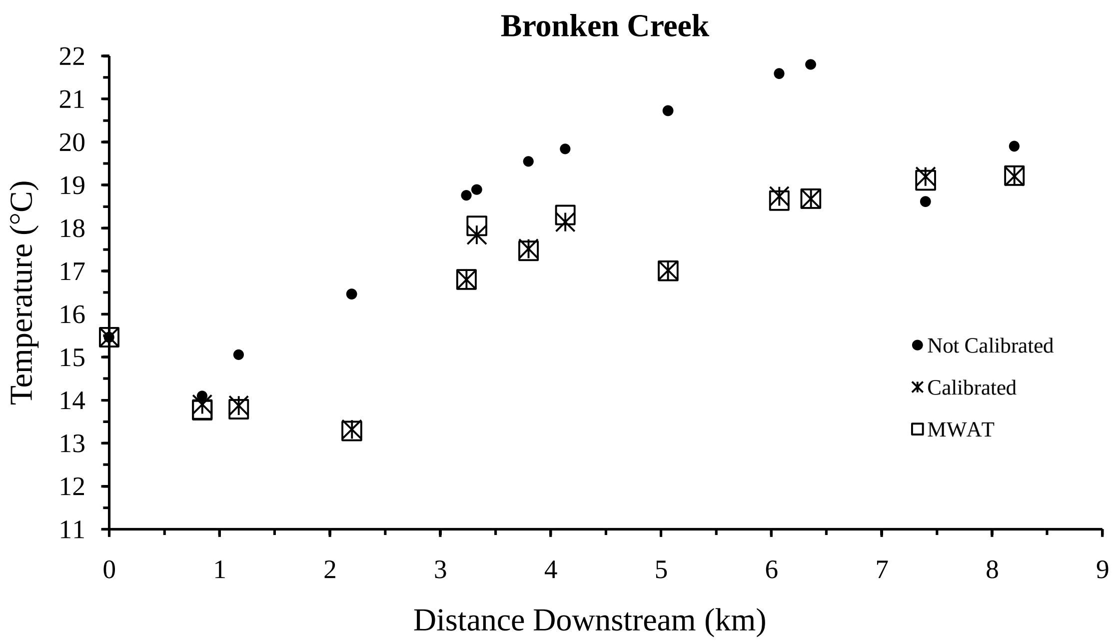



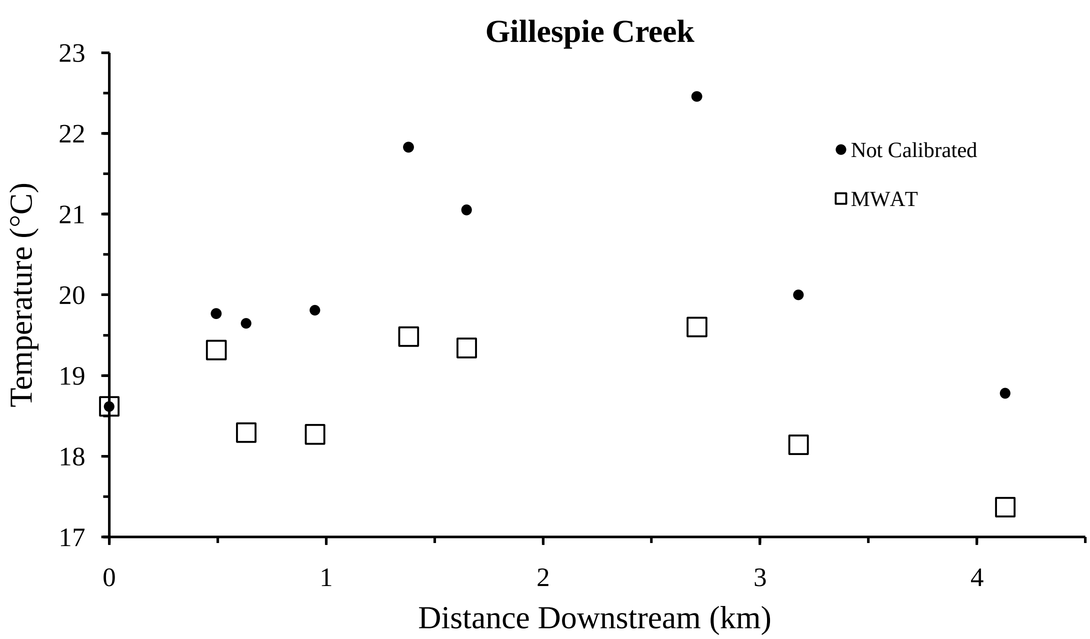
**Appendix B.** (Continued) MWAT compared to calibrated and non calibrated SSTEMP model predictions. Vertical lines represent locations of tributaries entering the streams.



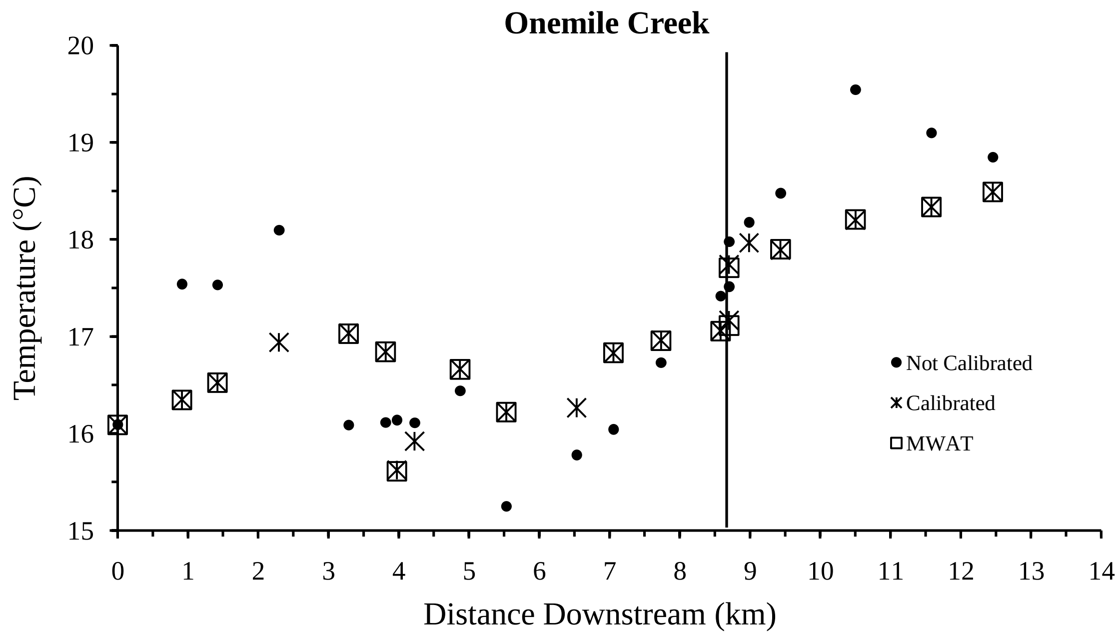



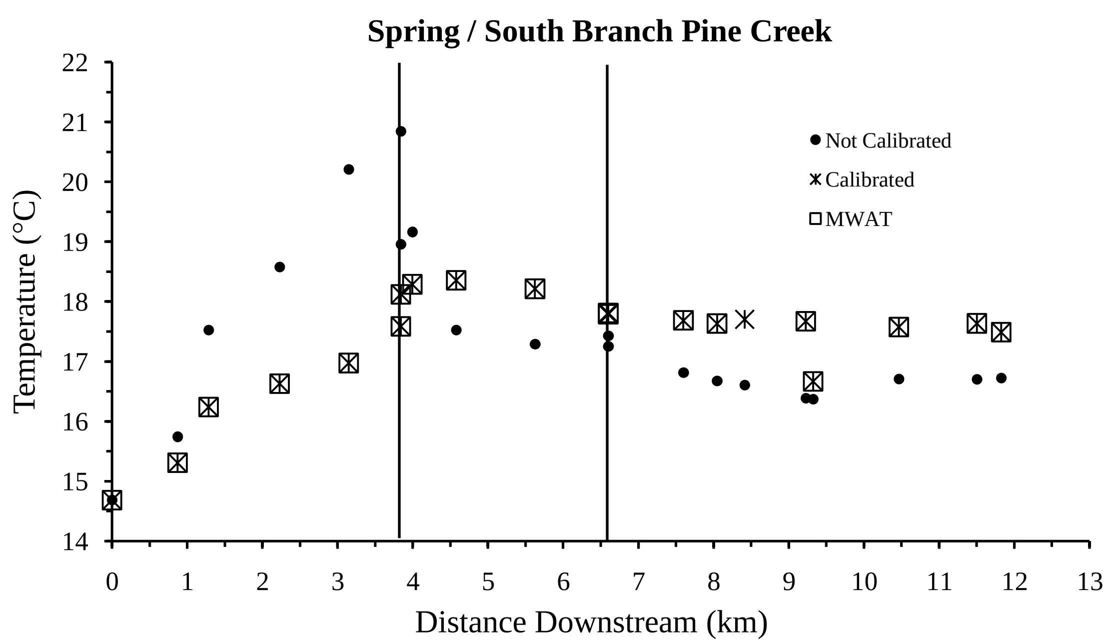
**Appendix B.** (Continued) MWAT compared to calibrated and non calibrated SSTEMP model predictions. Vertical lines represent locations of tributaries entering the streams.



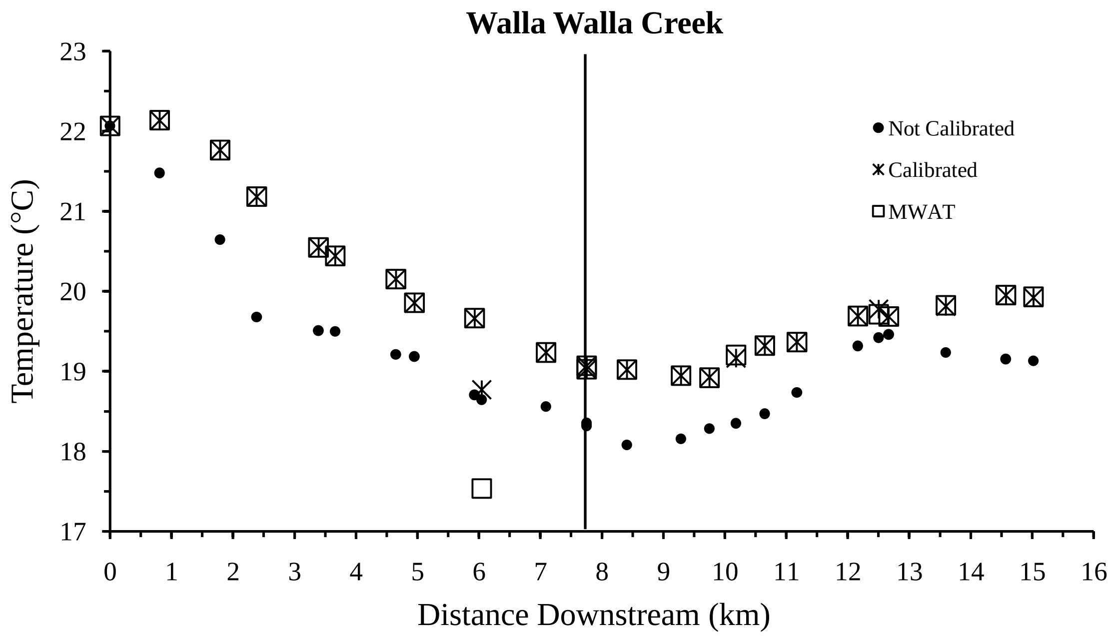



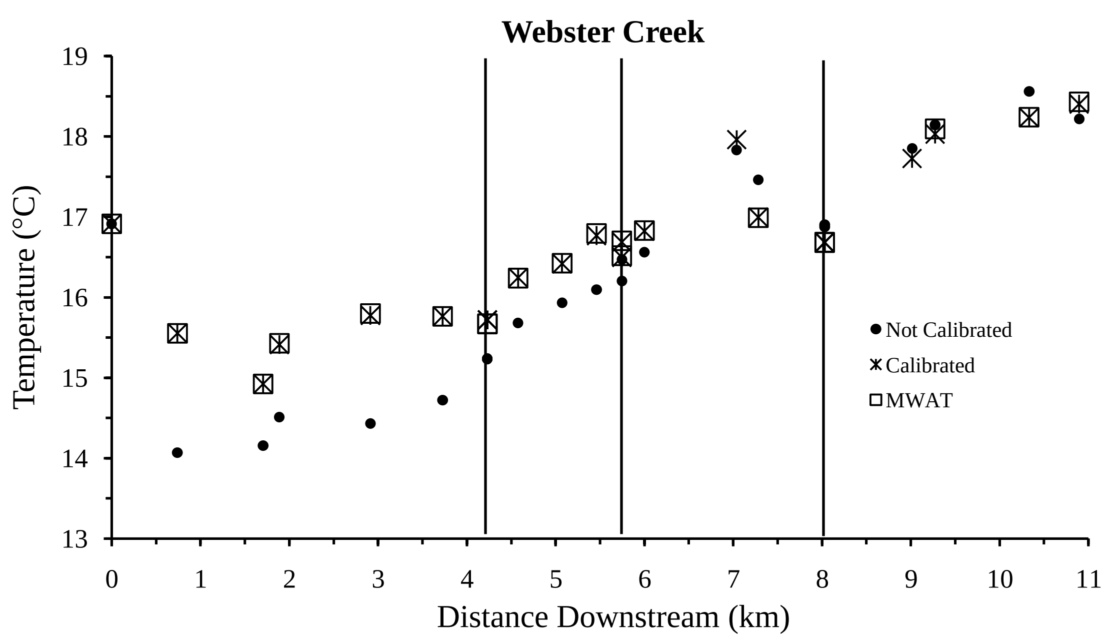
**Appendix B.** (Continued) MWAT compared to calibrated and non calibrated SSTEMP model predictions. Vertical lines represent locations of tributaries entering the streams.



**Appendix B.** (Continued) MWAT compared to calibrated and non calibrated SSTEMP model predictions. Vertical lines represent locations of tributaries entering the streams.



**Appendix C.** List of variables included in the multiple regression stepwise selection models.

Segment length (m)
Slope (m/m)
Latitude (°)
Upstream elevation (m)
Downstream elevation (m)
$\Delta$ Flow (cms) / stream length (m)
Stream Orientation (°)
Percent of segment with pool habitat (%)
Percent of segment with run habitat (%)
Percent of segment with riffle habitat (%)
LN Wetted width (m)
LN Flood prone width (m)
LN Bankfull width (m)
Maximum bankfull depth (m)
LN Entrenchment ratio (m/m)
Average stream depth (m)
Average stream thawlweg depth (m)
Topographic shade (°)
Percent riparian landuse cropland (%)
Percent riparian landuse developed (%)
Percent riparian landuse meadow (%)
Percent riparian landuse pasture (%)
Percent riparian landuse shrubland (%)
Percent riparian landuse woodland (%)
Percent riparian landuse wetland (%)
Bank angle (°)
Depth of undercut bank (m)
Shading vegetation offset (m)
Shading vegetation crown width (m)
Shading vegetation overhanging stream (m)
Average riparian buffer width out to 10 m (m)
Sand substrate (%)
Gravel substrate (%)
Silt substrate (%)
Boulder substrate (%)
Cobble substrate (%)
Shading vegetation shrubs (%)
Shading vegetation grasses (%)
Shading vegetation shrubs (%)
Air temperature (°C)
Stream shaded (%)
LN Vegetation height
LN Upstream flow (cms)
LN Downstream flow (cms)
LN Average flow (cms)
LN Width : Depth (m/m)

**Appendix D.** Descriptive statistics for variable used in multiple regression analysis.

| Variable | Range | Minimum | Maximum | Mean | Standard Error | Standard Deviation | Variance |
|---|---|---|---|---|---|---|---|
| Δ Flow/distance (cms/m) | 0.000160 | -0.000071 | 0.000089 | 0.000008 | 0.000001 | 0.000015 | 0.000000 |
| Upstream temperature | 9.81 | 13.28 | 23.09 | 18.8095 | 0.1898 | 2.1974 | 4.8285 |
| Sand substrate (%) | 1.00 | 0.00 | 1.00 | 0.7006 | 0.0237 | 0.2744 | 0.0753 |
| Segment length (m) | 1913.00 | 40.00 | 1953.00 | 751.4552 | 31.6085 | 365.8946 | 133878.8777 |
| Shade (%) | 93.25 | 1.25 | 94.50 | 44.8234 | 2.0410 | 23.6259 | 558.1823 |
| Slope (m/m) | 0.00522 | 0.00056 | 0.00579 | 0.00159 | 0.00009 | 0.0010 | 0.0000 |
| Upstream flow (cms) | 0.4149 | 0.0022 | 0.4171 | 0.1047 | 0.0088 | 0.1022 | 0.0104 |
| Wetted width (m) | 12.36 | 1.06 | 13.42 | 3.4978 | 0.1592 | 1.8428 | 3.3957 |
| Width (m) : depth (m) | 25.50 | 5.06 | 30.57 | 13.2988 | 0.4490 | 5.1971 | 27.0098 |

**Appendix E.** Results from multiple regression modeling to predict stream temperatures at the downstream end of each study segment during the period when MWAT occur using stepwise selection.

| Model | MWAT | Variable | | Intercept | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Downstream temperature (°C) | Coefficient | P | Coefficient | P | Adj. $R^2$ | AIC | AICc | SBC |
| 1 | Upstream temperature (°C) | 0.93822 | <0.0001 | 1.34737 | 0.0480 | 0.839 | -25.23 | -25.13 | -19.43 |
| 2 | Upstream temperature (°C) | 0.90867 | <0.0001 | 2.10837 | 0.0012 | 0.864 | -47.59 | -47.41 | -38.90 |
| | Δ Flow/distance (cms/m) | -24964.21570 | <0.0001 | | | | | | |
| 3 | Upstream temperature (°C) | 0.83532 | <0.0001 | 4.05953 | <0.0001 | 0.882 | -65.02 | -64.71 | -53.43 |
| | Δ Flow/distance (cms/m) | -28583.51398 | <0.0001 | | | | | | |
| | Slope (m/m) | -341.01672 | <0.0003 | | | | | | |
| 4 | Upstream temperature (°C) | 0.83507 | <0.0001 | 4.89185 | <0.0001 | 0.897 | -82.43 | -81.96 | -67.88 |
| | Δ Flow/distance (cms/m) | -23063.30849 | <0.0001 | | | | | | |
| | Slope (m/m) | -406.91123 | <0.0001 | | | | | | |
| | Sand substrate (%) | -1.09637 | <0.0001 | | | | | | |
| 5 | Upstream temperature (°C) | 0.81720 | <0.0001 | 4.81829 | <0.0001 | 0.905 | -91.60 | -90.94 | -74.21 |
| | Δ Flow/distance (cms/m) | -22784.06646 | <0.0001 | | | | | | |
| | Slope (m/m) | -419.38559 | <0.0001 | | | | | | |
| | Sand substrate (%) | -1.08755 | <0.0001 | | | | | | |
| | Segment length (m) | 0.00056 | 0.0011 | | | | | | |
| 6 | Upstream temperature (°C) | 0.79792 | <0.0001 | 5.27146 | <0.0001 | 0.910 | -99.31 | -98.42 | -79.03 |
| | Δ Flow/distance (cms/m) | -22304.04742 | <0.0001 | | | | | | |
| | Slope (m/m) | -362.08592 | <0.0001 | | | | | | |
| | Sand substrate (%) | -0.93035 | 0.0001 | | | | | | |
| | Segment length (m) | 0.00068 | 0.0001 | | | | | | |
| | Shade (%) | -0.00864 | 0.0025 | | | | | | |

**Appendix E.** (Continued)

| Model | MWAT<br>Downstream temperature (°C) | Variable<br>Coefficient | P | Intercept<br>Coefficient | P | Adj. $R^2$ | AIC | AICc | SBC |
|---|---|---|---|---|---|---|---|---|---|
| 7 | Upstream temperature (°C) | 0.76070 | <0.0001 | 4.78613 | <0.0001 | 0.916 | -106.78 | -105.06 | -83.60 |
| | Δ Flow/distance (cms/m) | -21358.00050 | <0.0001 | | | | | | |
| | Slope (m/m) | -324.33020 | <0.0001 | | | | | | |
| | Sand substrate (%) | -0.91031 | 0.0001 | | | | | | |
| | Segment length (m) | 0.00070 | 0.0002 | | | | | | |
| | Shade (%) | -0.01318 | <0.0001 | | | | | | |
| | LN Width (m) : depth (m) | 0.51486 | 0.0028 | | | | | | |
| 8 | Upstream temperature (°C) | 0.75122 | <0.0001 | 3.41483 | <0.0001 | 0.918 | -110.48 | -109.33 | -87.30 |
| | Δ Flow/distance (cms/m) | -26907.37202 | <0.0001 | | | | | | |
| | Slope (m/m) | -327.72774 | <0.0001 | | | | | | |
| | Shade (%) | -0.01281 | <0.0001 | | | | | | |
| | Segment length (m) | 0.00076 | <0.0001 | | | | | | |
| | LN Wetted width (m) | 0.91010 | <0.0001 | | | | | | |
| | LN Upstream flow (cms) | -0.40586 | <0.0001 | | | | | | |

**Appendix F.** Results for ANCOVA tests between-subjects effects for the change in stream temperature per km during the period of maximum daily average temperatures (MDAT). Parameter estimates are for one-way tests. Significant results are indicated by bolding.

| Variable | Effect | df | MSE | F | P |
|---|---|---|---|---|---|
| MDAT | Intercept | 1 | 18.714 | 3.79 | 0.0566 |
| Δ Temperature / km | Covariate | 1 | 16.962 | 3.44 | 0.0691 |
| | Vegetation Type | 2 | 18.090 | 3.67 | **0.0320** |

Parameter Estimates for one-way test

| Parameter | | Estimate | SE | T | P |
|---|---|---|---|---|---|
| Intercept | | 3.3453 | 2.15 | 1.56 | 0.1252 |
| Covariate | | -0.1942 | 0.10 | -1.85 | 0.0691 |
| Vegetation Type: | Grass | 1.7757 | 0.70 | 2.54 | 0.0140 |
| | Shrub | 0.4768 | 0.98 | 0.48 | 0.6302 |
| | Tree | 0.0000 | | | |

**Appendix G.** Results for ANCOVA tests between-subjects effects for the change in stream temperature per km during the period of maximum weekly average temperatures (MWAT). Parameter estimates are for one-way tests. Significant results are indicated by bolding.

| Variable | Effect | df | MSE | F | P |
|---|---|---|---|---|---|
| MWAT | Intercept | 1 | 10.442 | 2.745 | 0.1032 |
| Δ Temperature / km | Covariate | 1 | 9.601 | 2.524 | 0.1178 |
| | Vegetation Type | 2 | 11.608 | 3.052 | 0.0554 |

Parameter Estimates for one-way test

| Parameter | | Estimate | SE | T | P |
|---|---|---|---|---|---|
| Intercept | | 2.7523 | 1.99 | 1.38 | 0.1724 |
| Covariate | | -0.1705 | 0.11 | -1.59 | 0.1178 |
| Vegetation Type: | Grass | 1.3859 | 0.62 | 2.24 | 0.0294 |
| | Shrub | 0.1995 | 0.86 | 0.23 | 0.8184 |
| | Tree | 0.0000 | | | |

**Appendix H.** Results for ANCOVA tests between-subjects effects for the downstream temperature during the period of maximum weekly average temperatures (MWAT). Parameter estimates are for one-way tests. Significant results are indicated by bolding.

| Variable | Effect | df | MSE | F | P |
|---|---|---|---|---|---|
| Downstream | Intercept | 1 | 1.5921 | 2.740 | 0.1035 |
| Temperature | Covariate | 1 | 288.2232 | 496.105 | **<0.0001** |
| | Vegetation Type | 2 | 3.3539 | 5.773 | **0.0053** |

Parameter Estimates for one-way test

| Parameter | | Estimate | SE | T | P |
|---|---|---|---|---|---|
| Intercept | | 0.9975 | 0.78 | 1.28 | 0.2052 |
| Covariate | | 0.9340 | 0.04 | 8.71 | <0.0001 |
| Vegetation Type: | Grass | 0.7446 | 0.24 | -1.77 | 0.0033 |
| | Shrub | 0.1060 | 0.34 | -1.23 | 0.7549 |
| | Tree | 0.0000 | | | |

**Appendix I.** Results for ANCOVA tests between-subjects effects for the downstream temperature during the period of maximum daily average temperatures (MDAT). Parameter estimates are for one-way tests. Significant results are indicated by bolding.

| Variable | Effect | df | MSE | F | P |
|---|---|---|---|---|---|
| Downstream | Intercept | 1 | 1.7314 | 2.322 | 0.1333 |
| Temperature | Covariate | 1 | 399.6624 | 535.993 | **<0.0001** |
| | Vegetation Type | 2 | 4.6804 | 6.277 | **0.0035** |

Parameter Estimates for one-way test

| Parameter | | Estimate | SE | T | P |
|---|---|---|---|---|---|
| Intercept | | 0.8177 | 0.86 | 0.98 | 0.3318 |
| Covariate | | 0.9426 | 0.04 | 23.15 | <0.0001 |
| Vegetation Type: | Grass | 0.9286 | 0.27 | 3.42 | 0.0012 |
| | Shrub | 0.3561 | 0.38 | 0.93 | 0.3563 |
| | Tree | 0.0000 | | | |

**Appendix J.** Wisconsin trout stream thermal management decision making tree example for natural stream channels. Provides guidance on how to appropriately thermally manage trout streams and should be considered as of part of an overall trout management decision tree. Decreasing agriculture, forestry, and urbanization in watersheds and maintaining groundwater inputs are always appropriate management options when stream temperature is found to be a limiting factor.