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Stream temperature is a water quality parameter that directly influences the quality of aquatic habitat, particularly for cold-water species such as Pacific salmonids. Forest harvesting adjacent to a stream can increase the amount of solar radiation the stream receives, which can elevate stream temperatures and impair aquatic habitat. Oregon Forest Practice Rules mandate that forest operators leave Riparian Management Areas (RMAs) adjacent to streams in order to minimize the water quality impacts from forest harvesting. However, RMAs that contain overstory merchantable conifers are not required for small non-fish-bearing streams in Oregon, thus there is potential for increases in stream temperature to occur in headwater streams after harvesting. There is concern that increases in stream temperatures and changes to onsite processes in non-fish-bearing, headwater streams may propagate downstream and impair habitat in fish-bearing streams. The objectives of the following work are to assess the effects of contemporary forest management practices on stream temperatures of small non-fish-bearing headwater streams and to develop new knowledge regarding the physical processes that control reach-level stream temperature patterns.

Summer stream temperatures were measured for five years in six headwater streams in the Hinkle Creek basin in southern Oregon. After four years, four of the streams were harvested and vegetated RMAs were not left between the streams and harvest units. The watersheds of the two remaining

streams were not disturbed. Post-harvest stream temperatures were monitored for one year in all six streams. Each harvested stream was paired with one unharvested stream and regression relationships for maximum, minimum and mean daily stream temperatures were developed. Changes to temperatures of harvested streams were detected by comparing the mean pre-harvest regression relationship to the mean post-harvest relationship. Change detection analyses that considered the mean response among all four harvested streams indicated that maximum daily stream temperatures did not increase after harvesting, but that minimum and mean daily temperatures decreased significantly after harvesting. Additionally, diel stream temperature fluctuations were significantly greater one year after harvesting.

Pre- and post-harvest surveys of canopy closure in the harvested and unharvested streams were completed in order to compare levels of stream shading before and after harvest. The post-harvest survey quantified canopy closure from remaining overstory vegetation as well as from logging slash that partially covered the harvested streams. The surveys indicated that mean overstory canopy closure in the harvested streams decreased by 84% as a result of the harvest, but as the logging slash provided considerable cover, total canopy closure decreased by only 20%. It is possible that the logging slash effectively attenuated solar radiation and prevented extreme temperature increases in the harvested streams. However, it is likely that streamflow increased after harvesting and that the increased streamflow also prevented increases to maximum temperatures and contributed to lower minimum and mean stream temperatures.

The Influence of Contemporary Forest Harvesting on Summer Stream Temperatures in Headwater Streams of Hinkle Creek, Oregon

by
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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Kelly Maren Kibler, Author

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The influence of contemporary forest harvesting on summer stream temperatures in headwater streams of Hinkle Creek, Oregon

Chapter I: Introduction

Justification

Commercial forestry is a principal industry in Oregon and throughout the Pacific Northwest. Currently, Oregon has 28 million acres of land designated as forestland and 85,600 Oregonians are employed in the forestry industry (Oregon Forest Resources Institute 2006). The income generated and jobs supplied by the forestry industry are crucial to the economy of the state of Oregon. However, the forestlands of the Pacific Northwest support multiple uses in addition to timber, including recreation, high quality water resources, and habitat for terrestrial and aquatic wildlife. Intensive forestry operations may degrade the suitability of these lands to provide some beneficial uses. In an effort to minimize the environmental impact of commercial forestry on the landscape, the State of Oregon enacted the nation's first Forest Practices Act in 1971 to regulate forestland management. Since the Oregon Forest Practice Rules have been in effect, considerable resources have been directed to exploring procedures that lessen the impact of forest operations on Oregon's waterways while maintaining economically sustainable harvest practices.

In recent years, populations of native anadromous salmonids have been listed as federally Threatened or Endangered according to the national Endangered Species Act. Declines in populations of anadromous salmonids are correlated with habitat degradation associated with intensive forest management and stream temperature changes that occur in response to management of surrounding watersheds may adversely impact aquatic habitat for anadromous salmonids. However, the mechanisms and processes that influence reach-level stream temperature patterns are not completely understood and there is a need for data on the stream temperature effects of contemporary forest harvesting on privately owned, intensively managed forestland. The objectives of the following work are to

1. observe and quantify how stream temperatures in small, non-fish-bearing headwater streams respond to contemporary intensive harvesting practices, and
2. explain reach-level stream temperature responses through investigation of pre- and post-harvest canopy closure.

Literature review

Physical controls to stream temperature

Observed stream temperatures are the result of interactions between external sources of available energy and water and the in-stream mechanisms that respond to and distribute the inputs of energy and water from external sources (Poole and Berman 2001). Within Poole and Berman's categorization, external stream temperature drivers are defined as processes or conditions that control the relative amounts of energy and water that enter or leave a stream reach. Available incoming solar radiation and water from upstream, tributaries, or subsurface sources are examples of external stream temperature drivers. Conversely, characteristics inherent to the stream's physical structure and the near-stream environment exert an internal control on the stream temperature response to external inputs of heat and water. Stream shading, channel morphology, and substrate condition are examples of internal temperature controls.

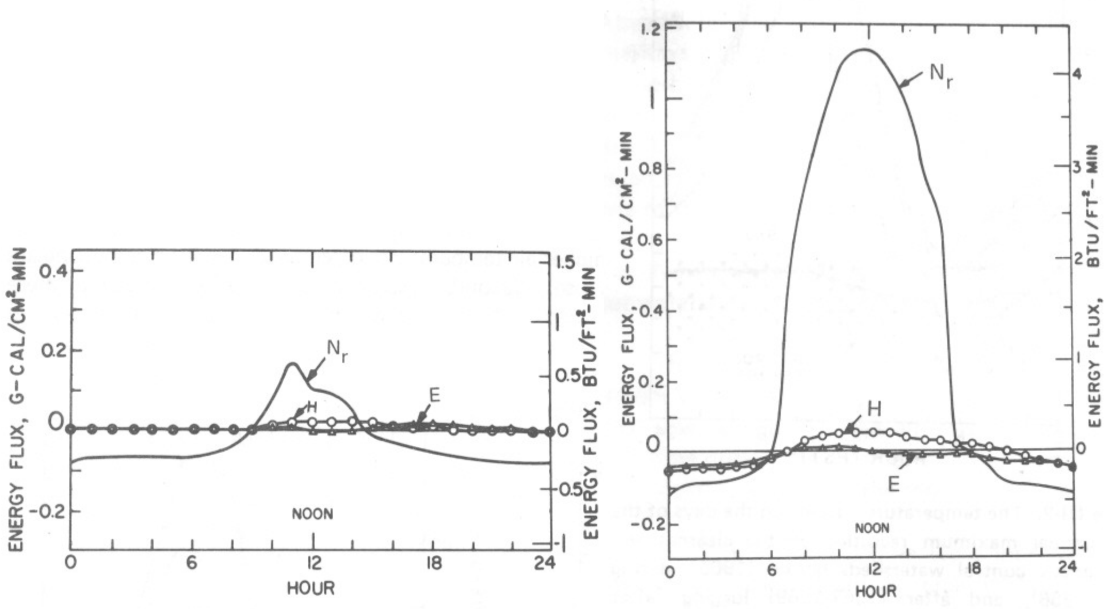
The sources of heat energy exchange between a stream and the surrounding physical environment can be summarized by the following model:

$$\Delta H = N \pm E \pm C \pm S \pm A$$

in which ΔH is the net heat energy gained or lost from the stream, N is heat exchanged by net radiation, E is heat exchange from evaporation or condensation, C is heat conducted between the stream water and substrate, S is heat convected between the stream water and air, and A is advection of incoming water from tributaries or subsurface sources (Moore et al. 2005, Johnson and Jones 2000). The net radiation term in the energy balance encompasses both inputs of shortwave (solar) and longwave (thermal) radiation less emissions of longwave radiation. The input of shortwave radiation is the only heat exchange process within the stream energy balance that is unidirectional; shortwave radiation is delivered to the stream in the form of solar energy but there is no mechanism for emission of shortwave radiation (Boyd and Kaspar 2003).

The primary external driver controlling stream temperature is the amount of solar radiation to which a stream is exposed (Brown 1969, Beschta et al. 1987, Johnson and Jones 2000, Johnson 2004). Brown's 1969 study demonstrated that temperature change in stream reaches that receive little to no advective input from groundwater sources can be predicted using an above ground energy balance approach. Within the energy balance, the incoming solar radiation term dominates the convective and evaporative components of the model, and thus has the greatest impact on the amount of energy available to the stream. Streams that are shaded, such as those that flow through intact forests and are covered by the canopy, receive less solar radiation than streams that are unshaded However radiation has the largest magnitude of any term in the energy balance model, even in a fully shaded stream (Figure 1.1).

The relative effect of available solar energy on stream temperature depends on the extent that solar radiation reaches the water surface. Material that shades the stream controls the amount of solar energy that reaches the stream surface by attenuating and reflecting solar radiation. Shade may be provided by over- or understory riparian vegetation in any stage of life or senescence. Topographic features or stream morphology and orientation may also affect a stream's exposure to solar radiation.



a b

Figure 1.1 Daily patterns of net radiation (N_r), evaporation (E) and convection (H) for a shaded (a) and unshaded (b) stream (Brown 1969).

The absolute amount of solar radiation that reaches a stream is only part of the mechanism by which stream temperatures are raised. The surface area and discharge of a stream are two additional factors that determine the extent to which the temperature of a stream will fluctuate in response to available solar radiation (Brown 1983). As the volume of water to be heated increases, the effect of a fixed amount of solar radiation becomes diluted and a smaller change in temperature is observed. Therefore, as stream discharge increases, the increase in stream temperature associated with a given amount of solar energy decreases. Conversely, as stream surface area increases, the amount of solar radiation that the stream can absorb also increases, which results in high net absorption per unit volume by a stream with a high surface area to volume ratio.

Some researchers have stated that convective heat exchange is a dominant process by which streams heat or cool (Larson and Larson 2001, Smith and Lavis 1975). However, because air temperature and solar radiation are highly correlated, it is often mistakenly concluded that air temperature controls stream heating when, in fact, it is radiative exchange driven by incoming solar radiation that causes stream temperature to increase (Johnson 2003). Energy balance analyses show that the magnitude of the incoming solar radiation term is considerably greater than the convective heat exchange term in the stream heat balance (Figure 1.1), (Brown 1969, Johnson and Jones 2000, Sinokrot and Stefan 1993).

Substrate type affects the way a stream absorbs solar energy. Johnson [2004] observed significant differences in maximum and minimum daily stream temperatures as well as daily stream temperature fluctuations when a bedrock reach was compared to an adjacent alluvial reach. Bedrock substrates of small, shallow streams can absorb radiant solar energy, thus becoming energy sources or sinks depending upon time of day. This process of absorption and storage can dampen the diel temperature signal by storing or releasing energy, resulting in lower maximum and higher minimum temperatures (Brown 1969). However, Johnson [2004] found that a bedrock reach had wider diel fluctuations than an alluvial reach, which suggests that the amount of solar energy absorbed by the bedrock during the day and released at night was not sufficient to dampen the diel fluctuation, as predicted by Brown [1969]. Furthermore, a dampening effect was observed after the stream flowed through the alluvial reach. The increased residence time of water within the alluvial reach may have allowed for conduction of heat between the surface water and the alluvial substrates, thereby cooling warmer water during the day and warming the cooler surface water at night.

Variable hydraulic residence times of individual streams may be instrumental in producing divergent temperature responses across streams that exhibit similar surface area to volume ratios and shade levels, and that are exposed to comparable levels of solar radiation. The degree that surface stream water interacts with the subsurface hyporheic zone can dramatically influence hydraulic residence times (Boulton et al. 1998, Morrice et al. 1997, Haggerty et al. 2002) and thus, temperature patterns within the surface water column (White et al. 1987). Streams characterized by high surface-hyporheic connection and long subsurface flowpaths may effectively thermoregulate through natural heat-exchange processes as warm surface water mixes with cooler subsurface water and remains in contact with subsurface alluvium (White et al.1987). Morrice et al. [1997] illustrated that hydraulic residence time increases with increasing hydraulic connection between surface flowpaths and the subsurface alluvial aquifer. Using both point-specific tracer analysis and reach-scale modeling, Morrice et al. [1997] demonstrated that surface-hyporheic interaction is controlled by hydrogeologic attributes of the channel substrate and the alluvial aquifer. Hydraulic conductivity of the substratum, the magnitude and orientation of hydraulic gradients, stream gradient and geomorphology and stream stage are physical variables that influence rates and volumes of surface-hyporheic exchange (Morrice et al. 1997, Haggerty et al. 2002). In streams examined by Morrice et al. [1997], substrates characterized by high hydraulic conductivities facilitated surface-hyporheic exchange, resulting in greater hydraulic residence times through a reach.

Though many studies and models agree that stream reach temperatures increase in response to land use activities that enhance a stream's exposure to solar radiation, there have been disparate conclusions to questions of downstream heat propagation and associated cumulative watershed impacts. With regard to an above-ground energy budget, the relatively diminutive magnitude of terms that could dispel heat (convection, conduction and evaporation) as compared to the incoming solar term is substantial. Solar radiation absorbed by a stream will result in an increase in stream temperature but the increase will not be easily dissipated by convection, conduction, and evaporation and therefore, theoretically, the stream will cool more slowly than it is heated (Brown 1983). There is ambiguity within current literature regarding what happens to stream temperature downstream of a reach that was warmed by inputs of solar radiation. Beschta and Taylor's [1988] thirty-year study of stream temperature and logging activity in the Salmon Creek watershed documents a significant relationship between stream temperature at the mouth of the watershed and cumulative harvesting effects which indicates that reach-level stream temperature increases are detectable downstream. Oregon Department of Forestry monitoring reports of the Brush Creek watershed indicate that stream temperatures heated as the stream flowed through a clearcut reach but then cooled so that there was no net heating observed at the watershed mouth (Robison et al. 1995, Dent 1997). A Washington study that focused on downstream effects of elevated temperatures in small streams concluded that temperature increases in small streams were mitigated within 150 meters of a confluence with a larger stream, however results varied from site to site (Caldwell et al. 1991). Finally, Johnson [2004] demonstrated that maximum temperatures in an exposed stream reach were cooler after the stream flowed through a 200-meter shaded section than before the stream entered the shaded section. The results of these studies signify that in some situations stream temperature downstream of a disturbance is able to recover somewhat more rapidly than is predicted by an above-ground energy balance but that the temperature response downstream of a heated reach is variable.

The primary process of energy dissipation within a stream is generally through evaporative heat flux, followed by emission of longwave radiation (Boyd and Kaspar 2003). While rates of longwave radiation emission are influenced only by water temperature, evaporative flux is controlled by conditions in the near-stream environment. Vapor pressure gradients at the air-water interface drive evaporation rates and so climatic conditions such as humidity and windspeed significantly affect rates of evaporative flux (Benner 1999, Boyd and Kaspar 2003, Dingman 2002). Gauger and Skaugset observed rates of evaporative heat flux on the order of 400 W/m^2 in a stream in the western Cascades of Oregon, and observed that wind enhanced rates of evaporative heat flux (Gauger and Skaugset, unpublished data). While most heat dissipation through evaporative heat flux occurs during the day when humidity gradients between the stream and air and wind speeds are greatest, net longwave emission away from the stream occurs at night when stream temperatures become warmer than air and sky temperatures.

Physical effects of stream temperature

Maximum annual stream temperatures lag nominally one to two months behind the time of annual maximum solar insolation (Beschta et al. 1987), however, the timing of maximum annual temperature may change when riparian vegetation is removed. Johnson and Jones [2000] report that streams with disturbed riparian canopies reached summer peak temperatures close to the time of maximum solar insolation despite the fact that stream discharge was still high at that time while nearby streams with undisturbed riparian canopies reached peak temperatures later in the summer. This observation reinforces the dominance of solar radiation in determining stream temperature.

Aquatic organisms utilize dissolved oxygen (DO) for respiration for at least a portion of their life cycle; thus DO concentration is a water quality parameter of high significance to aquatic ecosystem health and is regulated under the federal Clean Water Act. The solubility of oxygen decreases in water as temperature increases; thus DO concentrations decrease as water temperature increases. This relationship creates a direct link between water temperature and quality of aquatic habitat. DO is consumed as organic matter within the stream is oxidized by chemical and biological processes during decomposition (Berry 1975, Ice and Brown 1978). Decomposition of organic matter that is dissolved or suspended in the water column or associated with the stream benthos contributes to a stream's biological oxygen demand (BOD). Rates of leaching, decomposition and associated BOD increase as water temperature increases (Berry 1975). The addition of organic matter to headwater streams in the form of logging slash contributes significantly to the BOD of the system, dramatically reduces surface and intergravel DO concentrations and may cause fish stress and mortality (Moring and Lantz 1975, Berry 1975).

Streams depleted of DO reaerate as oxygen from the atmosphere diffuses into the water (Ice and Brown 1978). Reaeration through oxygen diffusion occurs at the water surface and is enhanced by turbulence of the water. Turbulence at the water-air interface entrains air into the water column and brings oxygen-depleted water to the surface where it can reaerate (Ice and Brown 1978). The rate of intergravel reaeration is low in comparison to surface reaeration because the rate of water flux through benthic sediments is much lower than stream velocities (Brown 1983, Berry 1975). Salmonids begin their life cycle in redds as eggs and alevins that inhabit interstitial spaces within streambed gravels and low intergravel DO levels can reduce their survival (Ringler and Hall 1975).

Ecological effects of stream temperature

Water temperature criteria for streams in the Pacific Northwest were developed to protect aquatic habitat for native, cold-water species, particularly salmonids (Sullivan et al. 2000). Anadromous salmonids spawn and rear in freshwater streams and resident salmonids fulfill their entire life cycles within freshwater streams (Everest 1987). Therefore, the thermal environment of a stream constitutes a vital metric of habitat quality that may determine the ability of a stream to support salmonid populations. A shift in thermal patterns of a stream may affect fish populations that are adapted to existing local conditions, either through direct physiological pathways or by indirectly modifying environmental conditions.

Stream temperatures that are sub-optimal can cause outright salmonid mortality or may impose nonlethal effects that influence salmonid growth, behavior (migration and reproduction) and pathogen resistance (Sullivan et al. 2000). The net effect of both lethal and nonlethal impacts to salmonid populations depends on a combination of the severity and duration of exposure to sub-optimal temperatures. Mortality occurs when either the threshold magnitude or duration of extreme temperature exposure is exceeded. Acute temperature effects include those that cause death after an exposure time of less than 96 hours. Water temperatures over 25˚C generally exceed maximum lethal temperature limits of salmonids (Brett 1952), although fish that have acclimated to warm temperatures may persist above this threshold for short periods of time (Brett 1956).

Chronic exposure to sublethal stream temperatures causes stress to salmonids that is manifested through multiple physiological and behavioral pathways and decreases the probability of salmonid survival (Elliot 1981, Sullivan 2000). Physiological responses to a range of elevated but sublethal temperatures indicate that while rates of some physiological functions such as metabolic rate and heart rate increase continuously with increasing temperature, other physiological functions such as growth rate and appetite increase with temperature to a specific threshold, beyond which function declines (Brett 1971). The development of a salmonid at the beginning of its life cycle from egg to alevin, to fry and smolt occurs entirely within freshwater streams and the rate of development at each life stage is largely controlled by stream temperature. Stream temperature controls embryonic growth rates, hatching time of embryos, time spent in the gravel of redds as alevin, and emergence times and growth rates of fry (Marr 1966, Brett 1969, Weatherley and Gill 1995). Growth rates of individual fry are determined by a balance of energy expended by metabolism, activity and excretion to energy obtained through food consumption. After basic survival demands are met, energy that remains is applied to growth and reproduction (Brett 1969, Sullivan et al. 2000). Brett [1969] related the variables of temperature and food consumption to growth rates of salmonid fry and determined that the optimum growth rate for all levels of food availability occurs at temperatures between 5-17˚C. Maximum growth rates occurred at 15˚C when excessive food was available, however temperatures for optimum growth decreased with decreasing food availability and no growth occurred at temperatures above 23˚C. Growth rates of fry influence survival and success in later life stages of development and may determine the amount of time a fry of an anadromous salmonid will spend in the stream before smolting and seaward migration occur (Quinn and Peterson 1996, Weatherley and Gill 1995).

Water temperature directly influences salmonid behavior. Salmonids may survive periods of exposure to sub-optimal temperatures by employing behavioral thermoregulation and physiological energy-saving mechanisms (Elliot 1981). Evidence of bioenergetic regulation of salmon fry in thermally stratified lakes demonstrates that although many physiological processes are maximized at 15˚C in the laboratory, under field conditions during times of low food availability, salmonids naturally prefer cooler ambient temperatures where maintenance metabolism is reduced (Brett 1971). Thermal heterogeneity within a stream occurs when cooler subsurface water enters the stream by subsurface seepage or hyporheic exchange, creating localized areas of cooler habitat relative to the ambient stream temperature. There is evidence that salmonids preferentially seek out thermal refugia during times of temperature stress. Increasing frequency of pockets of cooler water is positively correlated with increased salmonid abundance (Ebersole et al. 2003). Stream temperature also affects salmonid behavior during migrations and thermal barriers to spawning adults may influence spawning locations and migration timing (Lantz 1971).

An indirect effect of elevated stream temperature and increased radiation is higher productivity of the stream ecosystem and a corresponding increase in the availability of food, which has the potential to affect salmonid populations. While the direct relationships between stream temperature and salmonid health have been reasonably well observed and quantified through laboratory experiments, defining comparable magnitudes of influence through indirect pathways is a more challenging task due to the complexity of ecosystem-wide relationships and challenges of performing ecological research *in-situ* (Lee and Samuel 1976). In the Pacific Northwest, fish communities are the highest trophic echelon of instream biota, thus fish are indirectly influenced by changes in the productivity of lower trophic levels, which include input of allochthonous organic matter, instream primary production and aquatic invertebrates (Beschta et al. 1987). Water temperature directly affects chemical and biological processes that occur within the aquatic ecosystem, thus stream temperature is a ubiquitous control to the productivity of the stream ecosystem. Stream temperature influences rates of periphyton growth, organic matter decay and nutrient cycling by controlling rates of chemical transformations within the water column, (Berry 1975, Phinney and McIntire 1965). Increases in stream temperature and light availability that can result from forest harvesting may lead to shifts in biomass production, species composition and dominance of algal communities within the stream (Armitage 1980), which indirectly influences the trophic balance of the stream. Studies that compared in-stream productivity in harvested and unharvested streams often reported higher productivity in disturbed areas due to increases in light and temperature (Murphy and Hall 1981).

Indirect linkages between water temperature and salmonid health exist outside of the influence on food availability. The susceptibility of salmonids to disease and parasites increases in warmer temperatures, presumably due to the high metabolic rates and physiological stress associated with high temperatures (Ordal and Pacha 1963, Cairns et al. 2005). Stream temperature indirectly affects the quality of salmonid habitat by controlling the solubility of oxygen in stream water. Salmonid mortality caused by low DO concentrations occurs at concentrations less than 2mg/L, however nonlethal impacts to salmonids are observed at DO concentrations as high as 6mg/L (Hermann et al. 1962). Decreased growth rate, food consumption and food conversion (weight gain) were observed in juvenile coho salmon when DO concentrations decreased from 8.3 mg/L to 6 mg/L while mortality was observed at 2.3mg/L (Hermann et al. 1962).

Aquatic insects fill a vital niche in lotic ecosystems by processing organic material, thus providing a trophic link between primary production and higher tropic levels. The preponderance of evidence in scientific literature suggests that the instream thermal regime exerts a strong influence over the aquatic insect community. Although laboratory studies that tested the lethal limits of aquatic invertebrates showed that elevated or lowered water temperatures induced mortality when lethal limits of a given species are surpassed (Quinn et al. 1994), sublethal temperature effects may also influence the life history patterns and overall long-term survivability of macroinvertebrate populations. Water temperature affects the community structure of aquatic invertebrates (Gledhill 1960, Hawkins and Hogue 1997) and species extirpation was observed at temperatures above or below threshold temperatures (Sweeney 1978, Quinn et al. 1994, Nordlie and Arthur 1981, Sweeney and Schnack 1977). Peak macroinvertebrate densities and biomass occurred earlier in streams heated above ambient temperatures (Arthur 1982, Hogg and Williams 1996, Rogers 1980) and emergence of adult insects were observed earlier in streams heated as little as 2.5 to 3˚C above ambient temperatures (Nordlie and Arthur 1981, Hogg and Williams 1996, Rempel and Carter 1987). Stream temperature also influences rates of growth and affects reproductive success of aquatic insects. Temperature directly controls the metabolic rate of a given organism (Gillooly et al. 2001), and thus regulates the developmental rate of that organism (Rempel and Carter 1987) and directly affects mature body size (Hogg and Williams 1996, Sweeney and Vannote 1978, Sweeney and Schnack 1977). A compelling hypothesis that relates macroinvertebrate growth to the thermal environment states that each species has an optimal temperature regime that allows each individual to reach a maximum adult size and fecundity and that subjecting a species to a regime that is suboptimal (either warmer or cooler than optimal), results in reduced adult size and fecundity (Sweeney and Vannote 1978, Vannote and Sweeney 1980). This hypothesis is supported by data that demonstrate reduced adult body size for aquatic insects raised at temperatures above (Hogg and Williams 1996, Rempel and Carter 1987) and below (Sweeney and Schnack 1977, Sweeney and Vannote 1978, Sweeney 1978) the ambient thermal regimes as compared to populations raised within ambient temperatures and by studies correlating adult body size to fecundity (Rogers 1983, Sweeney and Vannote 1978, Hogg and Williams 1996).

Stream temperature and forestland management

The relationships between streamflow, solar radiation, shade and stream temperature are prominent in the Pacific Northwest, where intensively managed forest land and streams that support an economically, culturally and ecologically valuable salmon fishery coexist. Incoming solar radiation peaks during the summer months of May, June, July and August. Paradoxically, climate patterns in the Pacific Northwest result in low probabilities of rainfall and high probabilities of clear skies during the summer months, with the result that peak annual solar energy is available during the times of lowest annual stream discharge (Beschta et al. 1987). Small, headwater streams in the Pacific Northwest are vulnerable to increases in temperature during summer low flow months when incident solar radiation is high, particularly when riparian vegetation is removed from streams that were historically shaded by intact forest canopies.

Change to the thermal regimes of forest streams can be an undesirable effect of vegetation removal within the watershed. The historic Alsea Watershed Study demonstrated that the removal of streamside vegetation during forest harvesting caused increases in stream temperatures (Brown and Krygier 1970). Average monthly maximum stream temperatures increased 8˚C the summer after the forest adjacent to a small stream in Oregon’s Coast Range was clearcut. In the same stream, diel stream temperature range doubled after clearcutting. The importance of shade was further demonstrated in Levno and Rothacher’s [1967] work in the HJ Andrews Experimental Forest in western Oregon. Maximum weekly stream temperatures in a 96-hectare watershed that was clearcut harvested did not diverge significantly from pre-logging temperature patterns until 55% of the vegetation was removed from the watershed. In the same study, no significant changes to stream temperature patterns were observed one year after 25% of 101-hectare watershed was patch cut. Downed wood and understory vegetation remained near the stream in the patch-cut watershed the first year following harvesting, however this material was removed during a winter debris flow that scoured the channel to bedrock, exposing 1,300 feet of the channel to direct solar radiation. Stream temperatures were significantly higher following the debris flow than either before logging or one year after logging, which indicates that the downed vegetation provided shade to the stream and precluded stream temperature increases one year after logging. Brown and Krygier [1967] quantified a 9˚C increase in stream temperatures as water flowed through the 1,300-foot reach that had been was scoured.

The role of senescing organic material as a temporary agent of shade was defined in a study of headwater streams in western Washington (Jackson et al. 2001). Post-harvest stream temperatures in headwater streams were not significantly different than pre-harvest temperatures one year after the streams were clearcut without a vegetated buffer. Jackson et al. [2001] attributed the insignificant temperature response to the meter-thick layer of organic material (logging slash) that covered the clearcut streams and effectively excluded solar radiation after harvesting.

Increases to stream temperatures caused by forest harvest adjacent to streams can be mitigated by Best Management Practices (BMPs), such as retention of riparian vegetation on either side of a stream (Bescheta et al. 1987, Brown and Krygier 1970, Brazier and Brown 1973, Macdonald et al. 2003, Swift and Messer 1971). Gomi et al. [2006] reported increases in maximum daily stream temperature of 2-9˚C in unbuffered headwater streams while maximum daily temperatures in streams with 10- and 30-meter buffers did not increase significantly. Similarly, the temperature increases observed in the HJ Andrews and Alsea paired watershed studies occurred in streams where riparian vegetation was clearcut or removed by debris flows whereas the streams with intact riparian buffers did not warm significantly (Levno and Rothacher 1967, Brown and Krygier 1970).

The characteristics that optimize effectiveness of riparian buffers have been thoroughly studied are known. Brazier and Brown [1973] reported that the volume of commercial timber left in the riparian buffer did not correlate with the amount of energy deflected by the buffer but that the width of the buffer (up to 40 feet) and canopy density of the buffer was directly proportional to temperature protection. In an investigation of riparian temperature gradients and edge effects, Brosofske et al. [1997] concluded that a minimum buffer width of 45 meters was necessary to preserve an unaltered riparian microclimate. In addition to length, width and basal density considerations, the effectiveness of a buffer is directly related to its long-term stability. Macdonald et al. [2003] reported that windthrow often occurs in riparian buffers and the loss of canopy in years following harvesting inhibited stream temperature recovery.

To minimize the environmental effects of forest harvesting on streams, buffer rules were included in Oregon's Forest Practices Act (OFP). Current OFP regulations require forest operators to leave a buffer of riparian vegetation or a Riparian Management Area (RMA) adjacent to streams that support either populations of fish or a domestic use, or large and medium sized streams that do not support fish or a domestic water use. The width of the required RMA ranges from 6 to 30 meters from the stream, depending upon beneficial use (domestic, fish, or neither) and size classification (small, medium, large) of the stream. Within the RMA, forest operators are required to retain:

1. a Standard Target square footage of basal area per 300 meters of stream (basal area retention depends on stream use, stream size, and silvicultural system),
2. all understory vegetation within three meters of the high water level,
3. all overstory trees within six meters of the high water level,
4. all overstory trees that lean over the stream channel, and
5. a portion of live, mature conifer trees in the RMA (number of trees retained depends upon stream use and size) (Oregon Administrative Rule 629-635).

Rules regarding RMAs in other timber-harvesting states of the Pacific Northwest are similar to the buffer rules mandated in Oregon's Forest Practice

Rules. Like Oregon, California, Washington and Idaho designate varying RMA widths and canopy densities depending upon stream size and beneficial use (Adams 2007). Minimum RMA widths are greater for streams in Washington, Idaho and California than for streams in Oregon. Additionally, Washington designates a 15-meter core zone within the larger RMA for fish-bearing streams in which no harvesting may occur. Portions of non-fish-bearing streams in Washington, California, and Idaho that drain to fish-bearing streams are protected by required RMAs of merchantable timber. In Washington, the first 90-150 meters of perennial, non-fish-bearing stream above a confluence with a fish-bearing stream is protected by a no-harvest RMA while Idaho designates RMAs on the first 150-300 meters of non-fish-bearing stream above a confluence. California mandates that RMAs of overstory trees be retained on any stream that demonstrates aquatic life (Adams 2007). In Oregon, RMAs of overstory conifers are not required adjacent to small, non-fish-bearing streams that are not domestic water sources. OFP Rules may require that all understory vegetation and non-merchantable timber be retained within three meters of the stream depending on the Geographic Region in Oregon that the stream is located and the size of the watershed that the stream drains. In any case, small, non-fish-bearing streams are not afforded the protection of a vegetated RMA that is designated for larger streams.

There is concern that stream temperature increases that occur in these unbuffered headwater tributaries may propagate downstream to larger, fish-bearing reaches and that the combined impact of several warmed tributaries may degrade aquatic habitat in fish-bearing streams. Since the OFP Rules were first enacted, revisions have been made to update the Rules as the body of knowledge regarding the impacts of forest management has expanded. Recent recommendations by Oregon's Forest Practices Advisory Committee on Salmon and Watersheds (FPAC) include an extension of current buffer rules to include a 15-meter RMA on either side of the first 150 meters of small, non-fish-bearing streams above a confluence with a fish-bearing stream.

Within the 15-meter RMA, forest operators would be required to retain all non-merchantable timber as well as four square feet of basal area per 30 meters of stream. There is a need to determine what, if any, changes to stream temperature are observed in small, non-fish-bearing streams in response to current Forest Practice Rules and if impacts are observed, whether or not they warrant a change in the current legislation.

Chapter II: The influence of contemporary forest harvesting on summer stream temperatures in headwater streams of Hinkle Creek, Oregon

Introduction

Stream temperature is a physical water quality parameter that directly affects all aquatic life by controlling metabolism, growth, oxygen solubility, organic matter decomposition and nutrient cycling within the stream ecosystem (Phinney and McIntire 1965, Marr 1966, Brett 1969, Brett 1971, Berry 1975, Weatherley and Gill 1995). Changes to prevailing thermal regimes stimulate physiological and behavioral response mechanisms in aquatic biota and effects ranging from physiological stress, changes in growth rates, fecundity, trophic structure, competitive interactions and timing of life history events and mortality are observed ecosystem responses to changes in ambient water temperatures (Brett 1952, Brett 1971, Moring and Lantz 1975, Sweeney and Vannote 1978, Beschta et al. 1987, Hogg and Williams 1996). In extreme cases, changes to thermal characteristics may alter the stream environment to the extent that native species are no longer able to inhabit their historic range. Pacific salmonids are particularly vulnerable to increases in stream temperature as they are cold-water fishes with lethal thermal tolerance of approximately 25˚C that inhabit freshwater streams during almost every stage of their life cycle (Brett 1952).

Many interacting mechanisms and processes contribute to observed stream temperature patterns; however according to energy balance analyses, solar radiation exposure is the primary temperature determinant of small, shallow streams (Brown 1969, Johnson and Jones 2000, Johnson 2004). Solar radiation exposure is limited by shade, such as from an intact forest canopy, and extreme increases to reach-level stream temperatures have been observed when forest canopies are removed (Levno and Rothacher 1967, Brown and Krygier 1970, Swift and Messer 1971). Where Riparian Management Areas (RMAs) that include mature timber are used, some percentage of pre-harvest canopy closure is preserved and often significant changes to stream temperature are not observed (Levno and Rothacher 1967, Brown and Krygier 1970, Swift and Messer 1971, Macdonald et al. 2003, Gomi et al. 2006). Recently the role of logging slash as an agent of post-harvest shade has also been investigated. Jackson et al. [2001] attributed a damped post-harvest temperature response of clearcut streams to exclusion of solar radiation due to a thick layer of logging slash that was deposited over the streams.

A key focus of contemporary watershed management is the role of cumulative watershed effects from the summation of many seemingly benign individual activities that produce a significant additive effect (Beschta and Taylor 1988). Small, non-fish-bearing streams in some regions of Oregon do not require that RMAs of overstory conifers be left during forest harvesting and there is concern that reach-level stream temperature increases may propagate into cumulative watershed effects, affecting downstream salmonid habitat. In order to assess the likelihood of a cumulative watershed effect, it is important to understand processes and mechanisms of stream thermal dynamics operating at the reach scale. Considerable research has focused on the effects of forest harvesting on stream temperatures, however, much of the prominent research was done in the era of old growth conversion, using equipment and techniques that were replaced by modern practices and before the current suite of forest practice rules were put into place. An investigation of the effects of timber harvest on stream temperatures on privately owned, intensively managed forest land with young, harvest-regenerated forest stands harvested using contemporary forest practices is necessary to assess reach-level impacts of current practices.

The objectives of this study are to 1) identify and quantify changes that occur to stream temperatures directly downstream of harvested units the first summer after harvesting and 2) explain the stream temperature response by examining differences in solar radiation exposure pre- versus post-harvest. I hypothesize that the harvesting treatment will reduce canopy closure over the harvested streams and that the increased exposure to solar radiation will cause stream temperatures to become warmer after harvest.

Methods

Site description

This research was undertaken as part of the Hinkle Creek Paired Watershed Study in association with the Watersheds Research Cooperative. We examined the headwater streams of Hinkle Creek, a tributary to Calapooya Creek that drains into the Umpqua River. The Hinkle Creek basin is located in the western Cascades of southern Oregon, approximately 25 miles (40 kilometers) northeast of the city of Roseburg in Douglas County.

The Hinkle Creek watershed is comprised of two fourth-order stream basins, the North Fork (basin area 873 hectares) and the South Fork (basin area 1,060 hectares). The streams flow approximately southwest and northwest, respectively, before they reach a confluence at the western boundary of the study area. The elevation of the study area ranges from about 400 meters above mean sea level (msl) at the mouth of the watershed to about 1,250 meters above msl near the eastern boundary of the watershed. Mean annual precipitation ranges from 1,400 mm at the mouth of the watershed to 1,900 mm at the eastern divide.

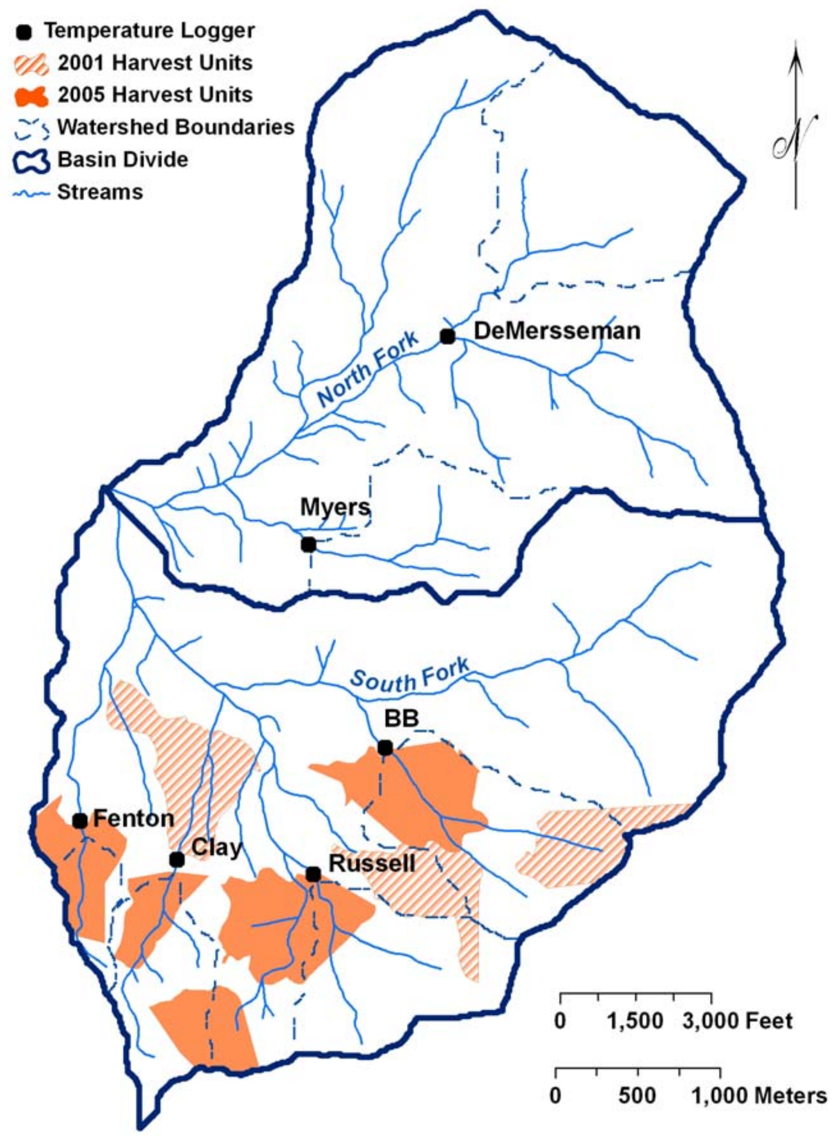


Figure 2.1 Hinkle Creek study area. Black points represent approximate locations of temperature data loggers, flumes, transition points between fish-bearing and non-fish-bearing streams and downstream limits to timber harvesting.

The vegetation in the Hinkle Creek basin is dominated by harvest regenerated stands of 55-year old Douglas fir (*Pseudotsuga menziesii*). Riparian vegetation is comprised of understory species such as huckleberry (*Vaccinium parvifolium*) and sword fern (*Polystichum munitum*), and overstory species such as red alder (*Alnus rubra*). The fish-bearing reaches of Hinkle

Creek contain resident cutthroat trout (*Oncorhynchus clarki*). Roseburg Forest Products (RFP) owns almost the entire watershed and the land is managed primarily for timber production. Before the commencement of the Hinkle Creek study in 2001, approximately 119 hectares of forest in the South Fork basin (11% of the South Fork Basin) was harvested in three clearcut harvest units (Figure 2.1).

Study design

The experimental design of the Hinkle Creek stream temperature study is a Before After Control Intervention (BACI) paired watershed study intended to identify and quantify stream temperature responses to forest harvesting in headwater streams. Six headwater watersheds were selected for study within the Hinkle Creek basin; four harvested (treatment) watersheds in the South Fork basin and two unharvested (control) watersheds in the North Fork basin (Figure 2.1). These headwater watersheds comprise the experimental units of the presented research and will be the focus of the following work. The orientation of the four treatment reaches in the South Fork basin is primarily south-north while the two control reaches in the North Fork basin flow approximately from west to east. Thirty-five hectares of the 2001 harvest units fell within the South Fork headwater watersheds investigated in this study. Four hectares (4%) of the Russell Creek watershed and 31 hectares (28%) of the BB Creek watershed were included in the 2001 harvest units (Figure 2.1). Each of the six headwater streams were instrumented with Montana flumes and stream temperature data loggers at the approximate transition point between a non-fish-bearing and fish-bearing stream designation so that stream reaches upstream of the flumes are designated as small, non-fish-bearing streams.

Harvesting treatment

Between July 2005 and March 2006, vegetation was harvested from the four South Fork watersheds while the watersheds of the North Fork remained unharvested. Harvest units were clearcut according to Oregon's Forest Practice Rules using modern harvesting techniques appropriate for each site. Most harvest units were yarded using a skyline logging system, however a portion of the harvest unit in the Fenton Creek watershed was shovel logged. Felled trees were yarded tree length to the landing where they were processed and removed from the project site via log trucks.

Table 2.1. Harvesting treatment. Areas of harvested and unharvested watersheds are shown in hectares (ha), total stream length within each watershed is given in meters (m), area of watershed harvested is given in hectares and percent of total watershed area, harvested stream length is given in meters and percent of total watershed stream length.

Watershed Name	**Harvested/ Unharvested Watershed**	**Area (ha)**	**Stream Length (m)**	**Area Harvested (ha, percent)**	**Harvested Stream Length (m, percent)**
Fenton Creek	Harvested	20	900	15, 75%	620, 69%
Clay Creek	Harvested	70	2,040	25, 36%	780, 38%
Russell Creek	Harvested	100	1,800	10, 10%	630, 35%
BB Creek	Harvested	110	2,280	35, 32%	1,060, 46%
Harvested Total		**300**	**7,020**	**85, 28%**	**3,090, 44%**
Myers Creek	Unharvested	90	2,100	-----	-----
DeMersseman Creek	Unharvested	160	1,580	-----	-----
Unharvested Total		**250**	**3,680**	**-----**	**-----**

The lower boundaries of the four harvest units coincided with the locations of Montana flumes, the point where the streams transitioned between a non-fish-bearing designation and a fish-bearing designation. Therefore, all stream reaches located within the harvest units were classified as small, non-fish-bearing reaches and according to the Oregon Forest Practice Rules, a Riparian Management Area (RMA) of merchantable timber was not required between the stream and harvest unit. Almost all merchantable timber and most non-merchantable timber and understory riparian vegetation was removed from riparian zones during harvesting. Logging slash, consisting of branches, needles and understory vegetation was left in place and harvested streams were partially covered by logging slash. Site preparation for replanting began in Spring 2006 and included herbicide treatments.

Stream temperature data collection

Summer stream temperatures in the six headwater watersheds were monitored over a four-year period of calibration data collection (2002 through 2005) followed by one year of post-harvest data collection (2006). Average stream temperature was recorded over 10 to 30 minute intervals using Vemco 12 bit Minlog data loggers (±0.2˚C accuracy, used 2002 and 2003), or HOBO Water Temp Pro data loggers (Onset HOBO model H20-001, ±0.2˚C accuracy, used 2004 through 2006). The data loggers were calibrated before deployment to ensure accuracy between locations. HOBO or Vemco data loggers were deployed each year in the late spring or early summer and continuously logged stream temperature data until late fall. Data loggers were located at the downstream edge of the proposed harvest units (Figure 2.1) and were placed in the same specific locations each year. During post-harvest data collection, data loggers were encased in white PVC covers to shade the instruments from direct solar radiation. Holes were drilled in the PVC cases to ensure that water flowed freely over the data loggers. Year-round stream temperatures were recorded within 10 meters of each seasonal data logger at 30 minute intervals (Campbell Scientific CS547A conductivity sensors ±0.1˚C accuracy, used November 2003 through 2006).

Canopy closure data collection

Surveys of canopy closure over the gauged streams were taken during the summer of 2004 and repeated during the summer of 2006. In this study, canopy closure is defined as the proportion of sky that is covered by vegetation that attenuates solar radiation before it reaches the stream (Jennings et al. 1999). The four harvested streams were surveyed at ten-meter intervals from a distance of 300 meters downstream of the downstream limit of the proposed harvest boundaries (flumes) to at least the upstream limits of the proposed harvest units (Figure 2.2). The unharvested streams were surveyed at ten meter intervals from a distance of 300 meters downstream from the flumes to at least 400 meters upstream of the flumes.

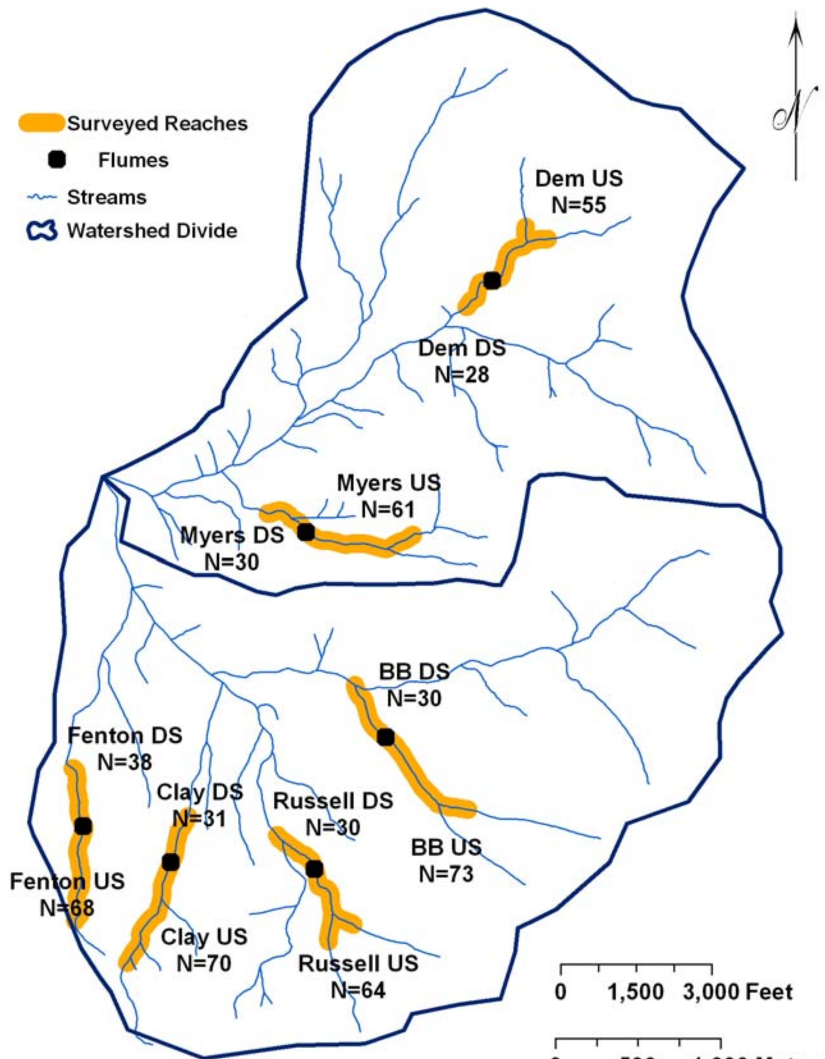


Figure 2.2. The locations of flumes and reaches surveyed for canopy closure in 2004 and 2006. The number of sampling points taken during the 2006 survey is displayed by each reach. The number of sampling points taken during the 2004 survey was equal or greater than the 2006 survey sample size for each reach.

Percent canopy closure was determined by measuring canopy closure upstream, downstream, perpendicular to the stream on river right and perpendicular to the stream on river left with a spherical densiometer held at waist height. The four canopy closure measurements at each location were averaged to calculate percent canopy closure at each sampling location. The densiometer operator took canopy closure measurements from the center of the stream.

During the summer of 2006, the percent canopy closure survey was repeated to gather post-harvest data on levels of shading in harvested and unharvested reaches. Percent canopy closure was sampled every ten meters along each of the six streams using methods similar to the pre-harvest survey. However, because the spherical densiometer held at waist height did not adequately characterize shade provided by downed vegetation in the streams, a second survey method was employed. Digital photos were taken at each sampling location from a perspective of two to eight inches above the water surface. Photos were taken close to the center of the stream at the exact location of densiometer data collection. A bubble level attached to the camera ensured that the photo captured a sampling area directly above the stream and each photo was taken facing north. The photos were analyzed by classifying proportions of light and dark pixels as canopy openness or closure, respectively in Adobe PhotoShop 7.0 software.

Data analysis

Maximum, minimum and mean daily stream temperatures

Parameter analysis of regression curves was used to detect changes to daily maximum, minimum and mean summer stream temperatures in Hinkle Creek (Meredith and Stehman 1991, Loftis et al. 2001). All statistical analysis was conducted within SAS version 9.1 (SAS Corporation, Cary, NC). Maximum, minimum and mean daily stream temperatures were extracted from the full temperature dataset of 10-30 minute observations and the three temperature metrics were analyzed separately. In order to meet the independence assumption inherent to regression, partial autocorrelation plots were examined for data from each stream, each year to determine the time period over which maximum daily temperatures were autocorrelated. This analysis indicated that the maximum lag time between autocorrelated values of daily maximum temperature was two days, thus a dataset consisting of the daily maximum temperature of every third day was systematically selected from the full dataset, with a randomly selected first day. Identical data selection techniques were used to select an independent set of minimum and mean daily temperatures. A two-day maximum lag time was identified for daily minimum and mean stream temperatures and so the final independent dataset also consisted of minimum and mean temperatures from every third day. Examination of residuals reflected that all assumptions of regression were adequately met by the data. Data from 2002 at Russell Creek were flawed due to direct solar absorption by the data logger and so data from this stream and year were removed from all analyses. Harvesting began in Fenton Creek during the summer of 2005, thus all stream temperature data collected in 2005 in Fenton Creek were not considered in this analysis.

A set of geographic and hydrologic characteristics for each watershed was considered to pair each harvested stream to an unharvested stream. Average basin aspect, average stream orientation, stream length upstream of the temperature sensors and stream discharge were considered in this analysis, resulting in the following stream pairings:

Table 2.2. Harvested-unharvested stream pairings for regression analysis.

Harvested Stream	Unharvested Stream	Pair Name
Fenton Creek	Myers Creek	Fen
Clay Creek	Myers Creek	Clay
Russell Creek	DeMerrseman Creek	Rus
BB Creek	DeMerrseman Creek	BB

After watershed pairing was established, the daily maximum temperatures from each harvested stream were plotted against daily maximum temperatures collected on the same day from the paired, unharvested stream. A Least Squares regression line was fit to data from each year, resulting in five regression lines (four pre-harvest and one post-harvest) for each stream pair, except for the Rus pair which lacked 2002 data from Russell Creek and the Fen pair which lacked 2005 data from Fenton Creek. From each regression line, a slope and intercept (˚C) parameter were extracted (Tables A1-A3). Before regression lines were fit to the paired harvested-unharvested relationships, the unharvested temperature data were adjusted by subtracting the mean value of the annual means of daily maximum temperature (2002-2006). This adjustment repositioned the scale of the x-axis, which allowed the intercept of the regression line to fall in the mid-range of the observed stream temperature values, precluding the need to extrapolate the intercept beyond the range of observed data. Similar regression analyses were performed for minimum and mean daily temperatures.

In order to detect changes between pre-harvest and post-harvest slopes and intercepts of the regression relationships, the following repeated measures model was fit to both the slope and intercept datasets:

$$\hat{\beta}_{\mathrm{ij}} = \mu_0 + \mathrm{S_j} + \mathrm{Y_i I_2} + \mathrm{Y_i I_3} + \mathrm{Y_i I_4} + \mathrm{Y_i I_5} + \varepsilon_{ij}$$

$\hat{\beta}_{\mathrm{ij}}$ = slope / intercept for year i (i = 2002, 2003, 2004, 2005, 2006), stream pair j (j = Fen, Rus, Clay, BB)

μ_0 = overall mean slope / intercept for all stream pairs, all years

$\mathrm{S_j}$ = random effect of stream pair that adds variability to the value of β, j = Fen, Rus, Clay, BB; $\mathrm{S_j} \sim \mathrm{N}(0, \sigma_{\mathrm{S}}^2)$

$\mathrm{Y_i}$ = effect of year i

$\mathrm{I_2}$ = indicator; = 1 if 2002, 0 otherwise

$\mathrm{I_3}$ = indicator; = 1 if 2003, 0 otherwise

$\mathrm{I_4}$ = indicator; = 1 if 2004, 0 otherwise

$\mathrm{I_5}$ = indicator; = 1 if 2005, 0 otherwise

ε_{ij} = random error term that represents variability between years;

$$\varepsilon'_{\mathrm{j}} \sim \mathrm{MN}(0, \Sigma_5) \quad \text{and } \Sigma_5 = \begin{matrix} 1 & \rho & \rho^2 & \rho^3 & \rho^4 \\ \rho & 1 & \rho & \rho^2 & \rho^3 \\ \rho^2 & \rho & 1 & \rho & \rho^2 \\ \rho^3 & \rho^2 & \rho & 1 & \rho \\ \rho^4 & \rho^3 & \rho^2 & \rho & 1 \end{matrix}$$

An autoregressive (AR(1)) correlation structure between time periods is the most appropriate correlation structure for repeated measures through time and therefore was selected for this model. Examination of residuals confirmed that the data adequately met all assumptions inherent to the model. Contrasts between mean slopes and intercepts before and after harvest were used to detect changes to the harvested-unharvested relationships of maximum, minimum and mean daily temperature that occurred between pre-harvest years and the post-harvest year.

Diel temperature fluctuation

Diel temperature fluctuation was calculated by subtracting the daily minimum temperature recorded at each stream from the daily maximum temperature. Diel ranges for every day between June 1 and September 30 were considered in this analysis. As diel range tends to fluctuate in a natural seasonal pattern throughout the summer, the season was divided into discrete periods and analyzed separately (Table 2.3).

Table 2.3. The warm season was divided into the following eight periods that were analyzed individually in the diel stream temperature analysis.

Period	Dates
1	June 1 to June 14
2	June 15 to June 30
3	July 1 to July 14
4	July 15 to July 31
5	August 1 to August 14
6	August 15 to August 31
7	September 1 to September 14
8	September 15 to September 30

Changes to diel range were detected by examining the diel range relationship between harvested and unharvested streams before and after harvesting. The pairing of harvested to unharvested streams employed in the maximum, minimum and mean analysis was also applied to diel analysis (Table 2.2). Missing data were simulated by interpolating within regression relationships between the HOBO temperature data logger at each site and the Campbell Scientific temperature probe located on the adjacent flume. The ratio of harvested to unharvested diel range was calculated for each stream pair and a repeated measures model was fit to the diel range ratio dataset. Examination of residuals indicated unequal variance, thus the natural log of the harvested to unharvested ratio of diel range was used to correct for heteroscadacity within the data. All other assumptions of the model were adequately met by the data. The following repeated measures model was used to detect changes to diel stream temperature fluctuation that occurred after harvesting:

$$\log(\hat{\beta}_{ij}) = \mu_0 + S_j + Y_i I_2 + Y_i I_3 + Y_i I_4 + Y_i I_5 + \varepsilon_{ij}$$

$\log(\hat{\beta}_{ij})$ = logged ratio of harvested over unharvested diel range for year i (i = 2002, 2003, 2004, 2005, 2006), stream pair j (j = Fen, Rus, Clay, BB)

μ_0 = overall mean ratio for all stream pairs, all years

S_j = random effect of stream pair that adds variability to the value of β, j = Fen, Rus, Clay, BB; $S_j \sim N(0, \sigma_S^2)$

Y_i = effect of year i

I_2 = indicator; = 1 if 2002, 0 otherwise

I_3 = indicator; = 1 if 2003, 0 otherwise

I_4 = indicator; = 1 if 2004, 0 otherwise

I_5 = indicator; = 1 if 2005, 0 otherwise

ε_{ij} = random error term that represents variability between years;

$$\varepsilon'_j \sim MN(0, \Sigma_5) \quad \text{and } \Sigma_5 = \begin{matrix} 1 & \rho & \rho^2 & \rho^3 & \rho^4 \\ \rho & 1 & \rho & \rho^2 & \rho^3 \\ \rho^2 & \rho & 1 & \rho & \rho^2 \\ \rho^3 & \rho^2 & \rho & 1 & \rho \\ \rho^4 & \rho^3 & \rho^2 & \rho & 1 \end{matrix}$$

An autoregressive (AR(1)) correlation structure between time periods is the most appropriate correlation structure for repeated measures through time and therefore was selected for this model. Contrasts between average diel ratio before and after harvest were used to detect changes to diel temperature range that occurred between pre-harvest years and the post-harvest year.

Greatest annual seven-day moving mean of the maximum daily temperature

Seven-day moving mean of the maximum daily stream temperature (seven-day mean) was calculated for every day of the summer for each stream, each year. The relationship of seven-day mean between harvested and unharvested streams was used to assess changes to seven-day mean that occurred after harvesting. The pairing of harvested to unharvested streams used in prior analyses was used to assess changes to annual maximum seven-day mean (Table 2.2). The maximum annual seven-day mean of each unharvested stream was subtracted from the maximum annual seven-day mean of the corresponding harvested streams. The following repeated measures model was used to assess changes to the differences between annual maximum seven-day means of harvested and unharvested streams after harvesting occurred:

$$\hat{\beta}_{\mathrm{ij}} = \mu_0 + \mathrm{S}_{\mathrm{j}} + \mathrm{Y}_{\mathrm{i}}\mathrm{I}_2 + \mathrm{Y}_{\mathrm{i}}\mathrm{I}_3 + \mathrm{Y}_{\mathrm{i}}\mathrm{I}_4 + \mathrm{Y}_{\mathrm{i}}\mathrm{I}_5 + \varepsilon_{ij}$$

$\hat{\beta}_{\mathrm{ij}}$ = difference between harvested and unharvested 7-day annual maximum for year i (i = 2002, 2003, 2004, 2005, 2006), stream pair j (j = Fen, Rus, Clay, BB)

μ_0 = overall mean difference for all stream pairs, all years

S_{j} = random effect of stream pair that adds variability to the value of β, j = Fen, Rus, Clay, BB; $\mathrm{S}_{\mathrm{j}} \sim \mathrm{N}(0, \sigma_{\mathrm{S}}^{2})$

Y_{i} = effect of year i

I_2 = indicator; = 1 if 2002, 0 otherwise

I_3 = indicator; = 1 if 2003, 0 otherwise

I_4 = indicator; = 1 if 2004, 0 otherwise

I_5 = indicator; = 1 if 2005, 0 otherwise

ε_{ij} = random error term that represents variability between years;

$$\varepsilon'_{\mathrm{j}} \sim \mathrm{MN}(0, \Sigma_5) \quad \text{and } \Sigma_5 = \begin{matrix} 1 & \rho & \rho^2 & \rho^3 & \rho^4 \\ \rho & 1 & \rho & \rho^2 & \rho^3 \\ \rho^2 & \rho & 1 & \rho & \rho^2 \\ \rho^3 & \rho^2 & \rho & 1 & \rho \\ \rho^4 & \rho^3 & \rho^2 & \rho & 1 \end{matrix}$$

An autoregressive (AR(1)) correlation structure between time periods is the most appropriate correlation structure for repeated measures through time and therefore was selected for this model. Examination of residuals confirmed that the data adequately met all assumptions inherent to the model. Post-harvest differences between harvested and unharvested seven-day means were compared to the mean pre-harvest differences using contrasts.

Cumulative degree days

A qualitative comparison of cumulative degree days was undertaken for each stream for years 2004, 2005 and 2006. Cumulative degree days (˚C) from March 1 to September 30 were calculated using mean daily temperature and were plotted for each harvested stream and one unharvested stream.

Canopy closure

Mean percentages of canopy closure and standard deviations from the mean were calculated for each reach (US = upstream of flumes and DS = downstream of flumes) of harvested and unharvested streams for the 2004 and 2006 canopy closure surveys and for both data collection method used during the 2006 survey. Differences between mean percentages of canopy closure recorded in unharvested reaches (Myers US, Myers DS, DeMerrseman US, DeMerrseman DS, Fenton DS, Russell DS and BB DS) were used to estimate the errors between different field crews using the densiometer method and errors between the densiometer and photo methods. Because the Clay DS reach was harvested in 2001 before the onset of the project, data from this reach do not represent unharvested values and thus were not included in the error analysis.

Results

Maximum, minimum and mean daily stream temperatures

Stream temperatures observed in harvested streams were highly correlated to data observed in unharvested streams during the calibration and post-harvest periods of data collection. Most stream pairs exhibited adjusted R^2 values of over 0.95 for maximum, minimum and mean daily temperatures for all years of data collection (Table 2.4). Slope and intercept parameters for all regression lines are in Tables A1-A3 in Appendix A.

Table 2.4. A list of correlation coefficients between maximum, minimum and mean daily stream temperatures for every third day in harvested and unharvested streams.

Stream Pair	Year	Maximum Daily Stream Temperature Adjusted R^2	Minimum Daily Stream Temperature Adjusted R^2	Mean Daily Stream Temperature Adjusted R^2
Fen	2002	0.96	0.94	0.97
Fen	2003	0.98	0.98	0.99
Fen	2004	0.97	0.99	0.99
Fen	2006*	0.92	0.94	0.96
Clay	2002	0.94	0.99	0.99
Clay	2003	0.94	0.99	0.99
Clay	2004	0.98	0.98	0.97
Clay	2005	0.99	0.99	0.99
Clay	2006*	0.91	0.97	0.97
Rus	2003	0.94	0.95	0.95
Rus	2004	0.95	0.96	0.97
Rus	2005	0.98	0.99	0.99
Rus	2006*	0.98	0.98	0.98
BB	2002	0.89	0.96	0.98
BB	2003	0.97	0.96	0.97
BB	2004	0.96	0.96	0.98
BB	2005	0.97	0.99	0.99
BB	2006*	0.97	0.97	0.98

*post-harvest

Statistically significant changes to the maximum daily stream temperature relationship between harvested and unharvested streams were not detected following harvesting at Hinkle Creek (Tables 2.5a and 2.5b, Figures 2.3a and 2.4a). Additionally, significant changes to intercepts of regressions on minimum and mean daily temperatures were not detected (Tables 2.5d and 2.5f, Figures 2.3b, 2.3c, 2.4b and 2.4c); however, post-harvest slopes of minimum and mean daily temperature regressions were significantly lower than pre-harvested slopes (minimum: $t_{10} = 8.64$, $p < 0.0001$, Table 2.5c, Figures 2.3b and 2.4b; mean: $t_{10} = 6.45$, $p < 0.0001$, Table 2.5e, Figures 2.3c and 2.4c). Slopes of post-harvest regressions of minimum daily temperature decreased by 0.26 relative to pre-harvest slopes (95% CI: 0.20 to 0.33) and slopes of post-harvest regressions on mean daily temperature decreased by 0.20 (95% CI: 0.13 to 0.27). Tables 2.5a-2.5f outline the differences in pre-harvest and post-harvest slopes and intercepts of regressions of maximum, minimum and mean daily temperatures for each individual stream pair as well as overall means.

Table 2.5a: Differences between pre-harvest mean slopes and post-harvest slopes of daily maximum stream temperature regressions for each individual stream pair and overall.

Stream Pair	Pre-Harvest Mean Slope (2002 to 2005)	Post-Harvest Slope (2006)	Change in Slope (Post-Pre)
Fen	0.92	0.64	-0.28
Clay	1.27	1.27	0.00
Rus	1.16	1.17	0.01
BB	0.82	1.11	0.30
Mean Slope	1.04	1.05	0.01

Table 2.5b: Differences between pre-harvest mean intercepts and post-harvest intercepts of daily maximum stream temperature regressions for each individual stream pair and overall.

Stream Pair	Pre-Harvest Mean Intercept (2002 to 2005)	Post-Harvest Intercept (2006)	Change in Intercept (Post-Pre)
Fen	13.68	12.11	-1.57
Clay	14.11	15.22	1.11
Rus	12.06	12.66	0.60
BB	12.89	13.64	0.75
Mean Intercept	13.19	13.41	0.22

Table 2.5c: Differences between pre-harvest mean slopes and post-harvest slopes of daily minimum stream temperature regressions for each individual stream pair and overall.

Stream Pair	Pre-Harvest Mean Slope (2002 to 2005)	Post-Harvest Slope (2006)	Change in Slope (Post-Pre)
Fen	0.91	0.59	-0.32
Clay	1.28	1.08	-0.20
Rus	1.28	0.98	-0.30
BB	1.34	1.05	-0.29
Mean Slope	1.19	0.93	-0.26

Table 2.5d: Differences between pre-harvest mean intercepts and post-harvest intercepts of daily minimum stream temperature regressions for each individual stream pair and overall.

Stream Pair	Pre-Harvest Mean Intercept (2002 to 2005)	Post-Harvest Intercept (2006)	Change in Intercept (Post-Pre)
Fen	12.78	10.93	-1.85
Clay	12.95	12.36	-0.59
Rus	11.31	10.39	-0.38
BB	12.08	12.09	0.01
Mean Intercept	12.28	11.58	-0.70

Table 2.5e: Differences between pre-harvest mean slopes and post-harvest slopes of mean daily stream temperature regressions for each individual stream pair and overall.

Stream Pair	Pre-Harvest Mean Slope (2002 to 2005)	Post-Harvest Slope (2006)	Change in Slope (Post-Pre)
Fen	0.92	0.62	-0.30
Clay	1.28	1.18	-0.10
Rus	1.26	1.06	-0.20
BB	1.32	1.10	-0.22
Mean Slope	1.19	0.99	-0.20

Table 2.5f: Differences between pre-harvest mean intercepts and post-harvest intercepts of mean daily stream temperature regressions for each individual stream pair and overall.

Stream Pair	**Pre-Harvest Mean Intercept (2002 to 2005)**	**Post-Harvest Intercept (2006)**	**Change in Intercept (Post-Pre)**
Fen	13.24	11.53	-1.38
Clay	13.49	13.72	0.24
Rus	11.70	11.66	-0.04
BB	12.48	12.79	0.31
Mean Intercept	12.73	12.42	-0.31

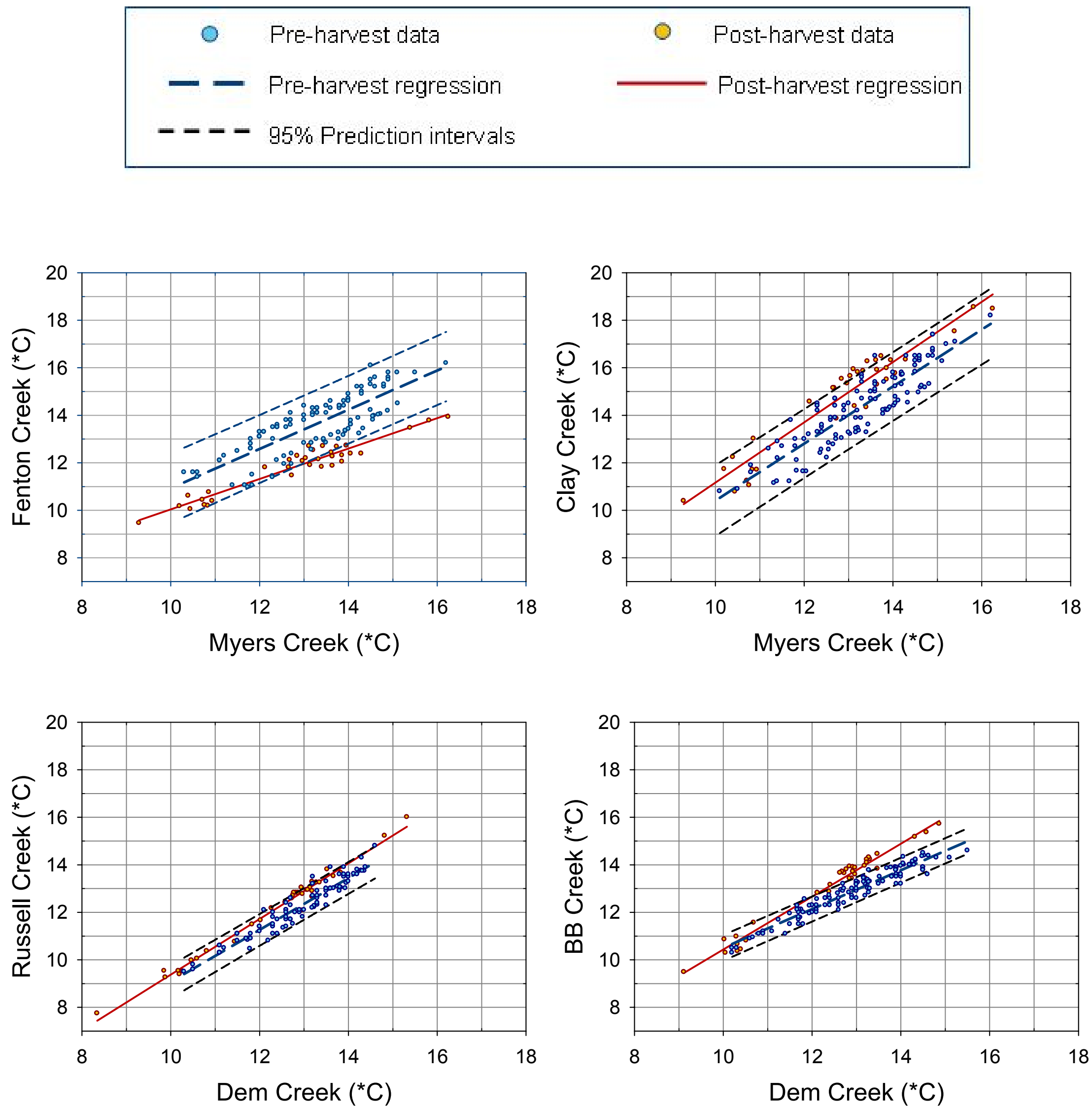


Figure 2.3a. Regressions of maximum daily stream temperatures in harvested streams versus unharvested streams. Each stream pair is shown individually. 95% prediction limits are around pre-harvest data.

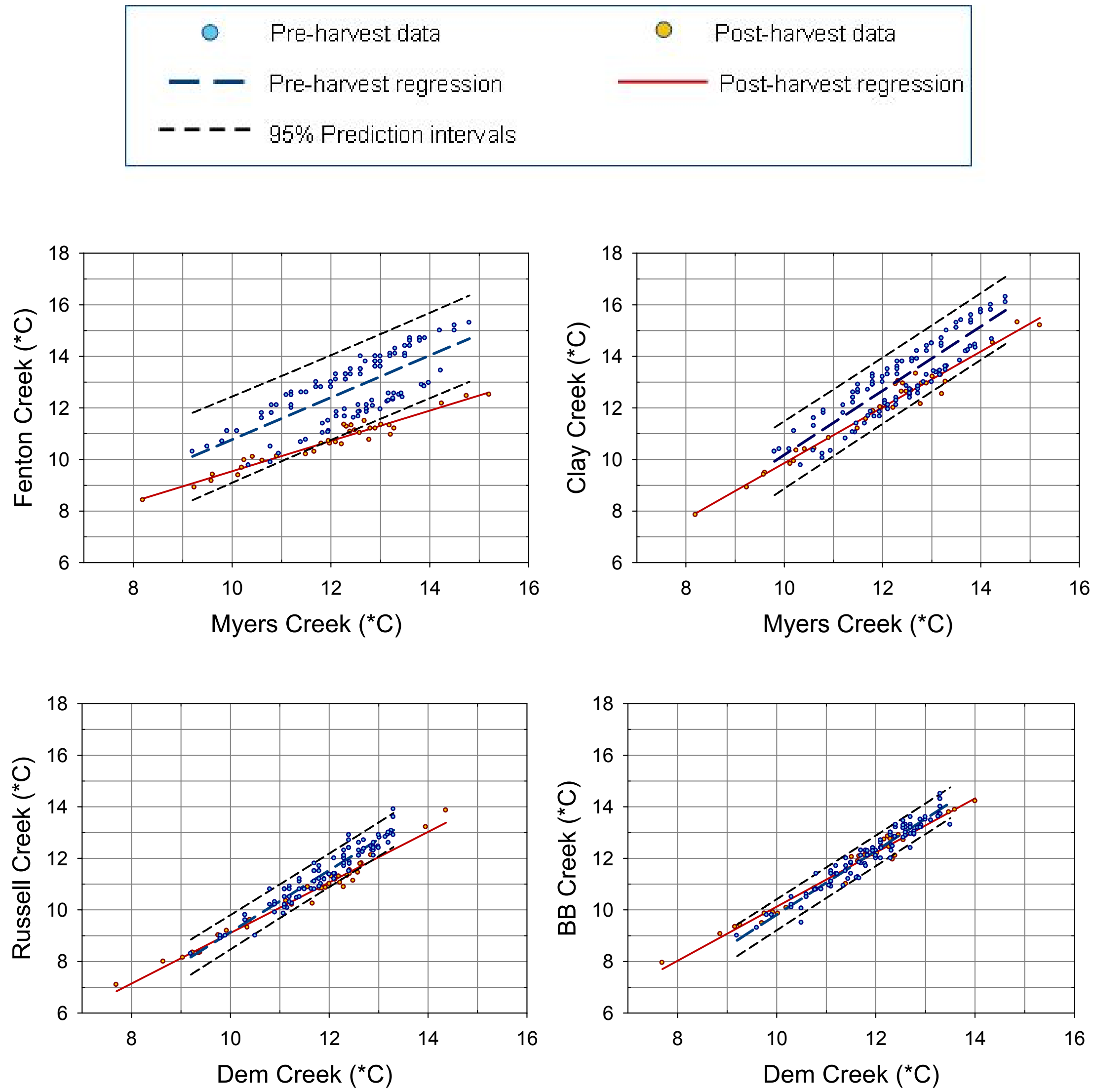


Figure 2.3b. Regressions of minimum daily stream temperatures in harvested streams versus unharvested streams. Each stream pair is shown individually. 95% prediction limits are around pre-harvest data.

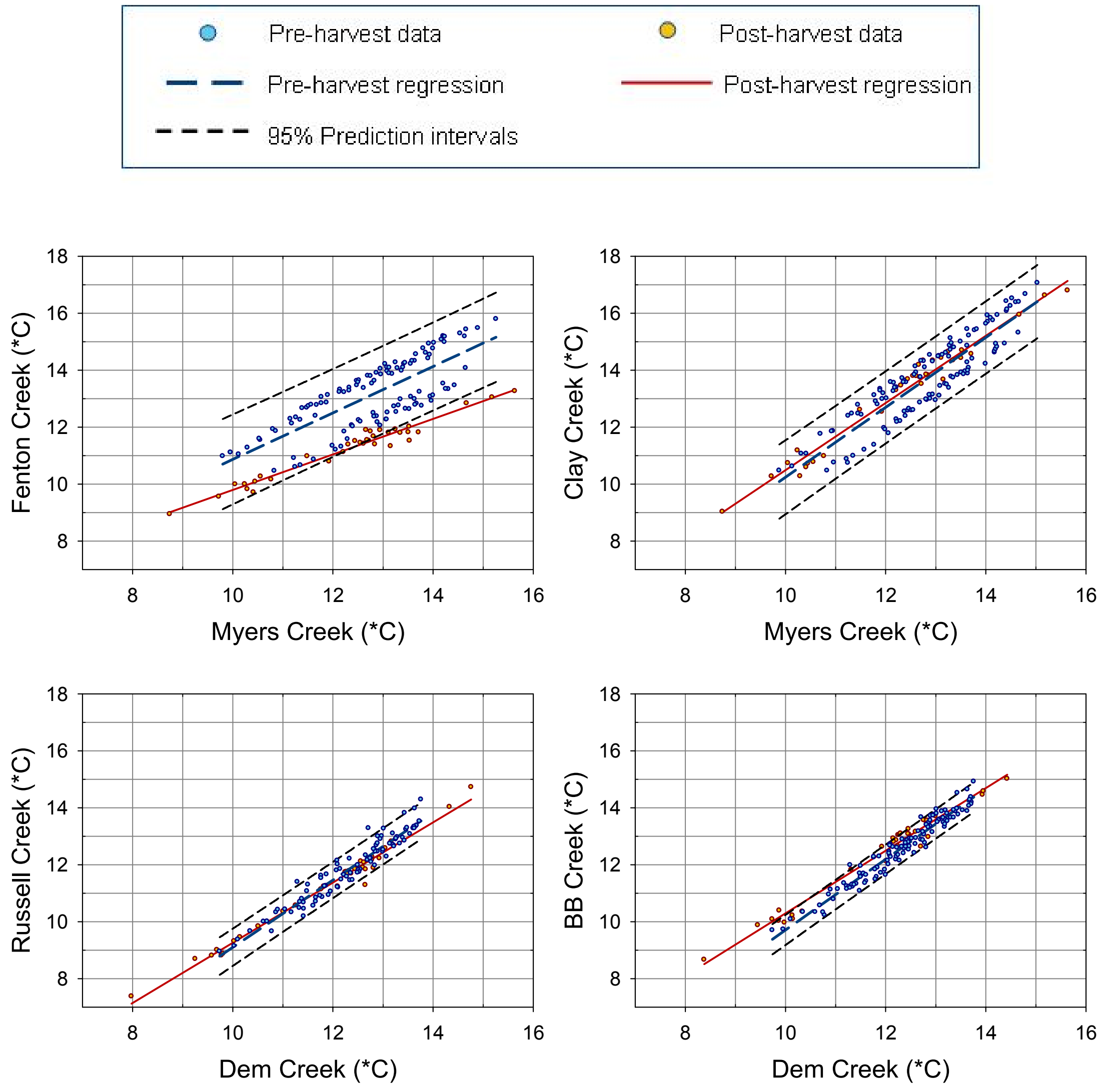


Figure 2.3c. Regressions of mean daily stream temperatures in harvested streams versus unharvested streams. Each stream pair is shown individually. 95% prediction limits are around pre-harvest data.

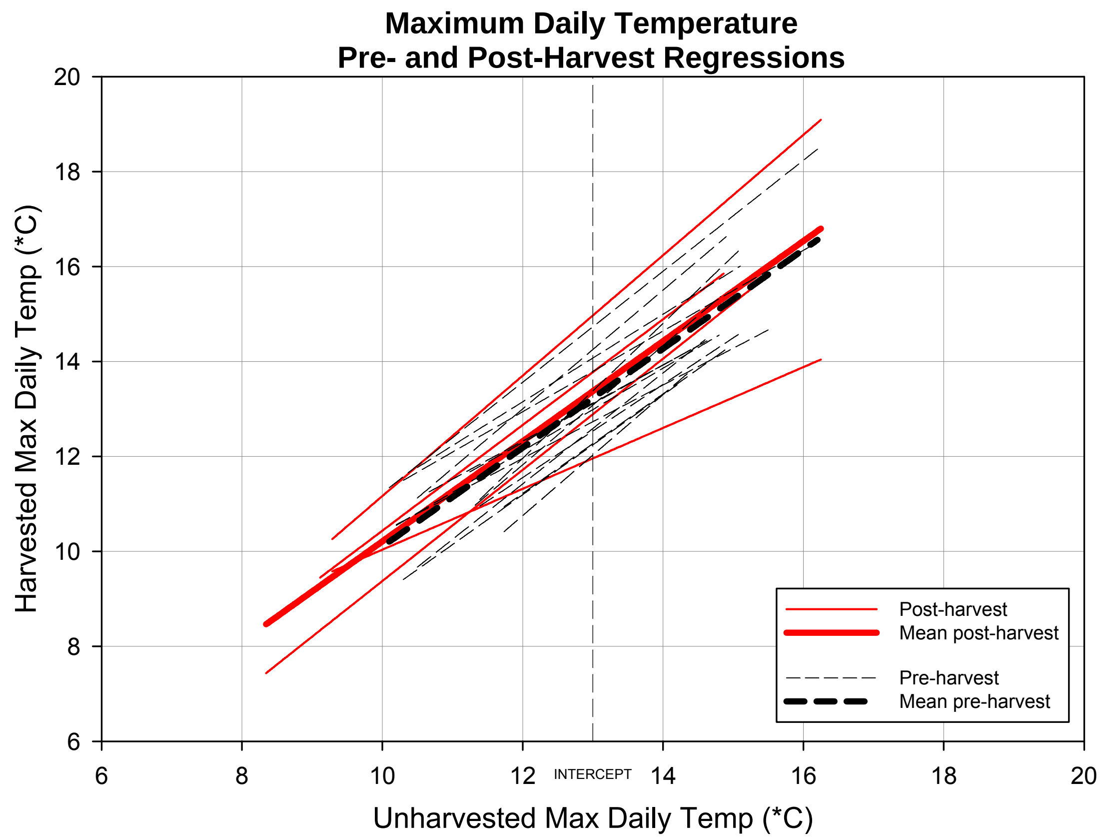


Figure 2.4a. Regressions of maximum daily stream temperatures in harvested streams versus unharvested streams for each stream pair and year illustrate variability of the harvested-unharvested relationship before and after harvest. Mean pre- and post-harvest regressions illustrate comparisons made by the change detection model. Vertical dashed line indicates mean intercept.

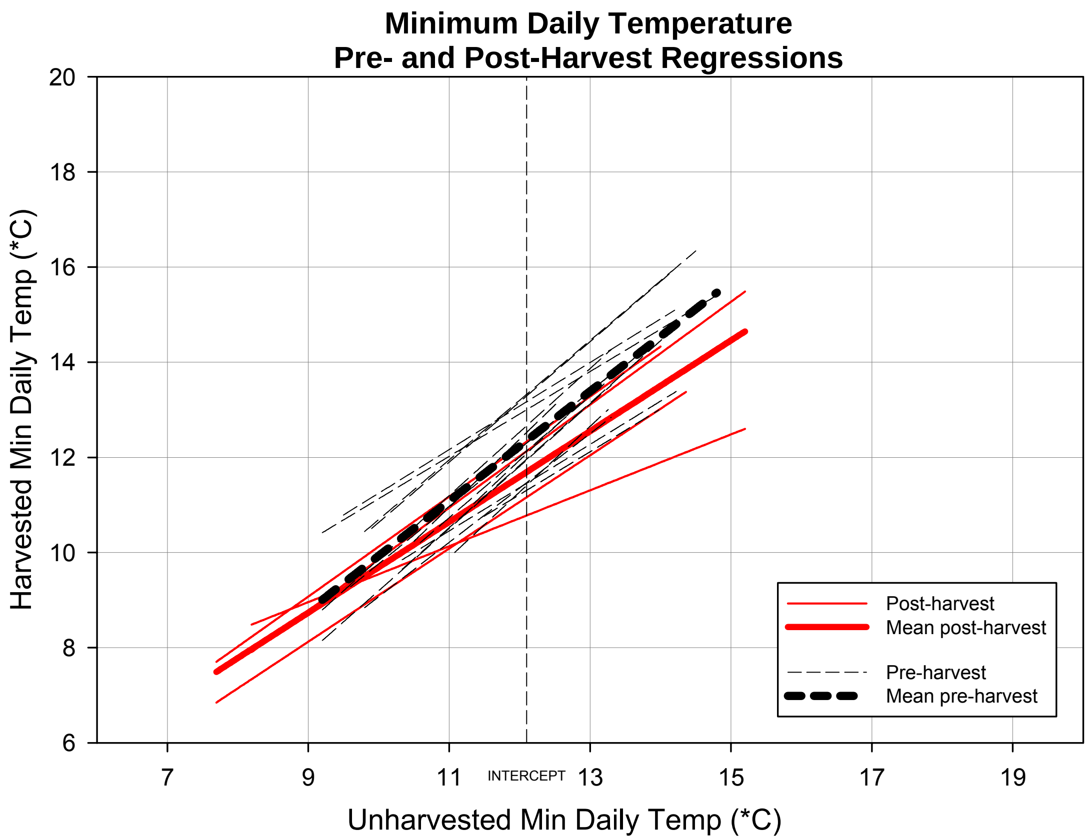


Figure 2.4b. Regressions of minimum daily stream temperatures in harvested streams versus unharvested streams for each stream pair and year illustrate variability of the harvested-unharvested relationship before and after harvest. Mean pre- and post-harvest regressions illustrate comparisons made by the change detection model. Vertical dashed line indicates mean intercept.

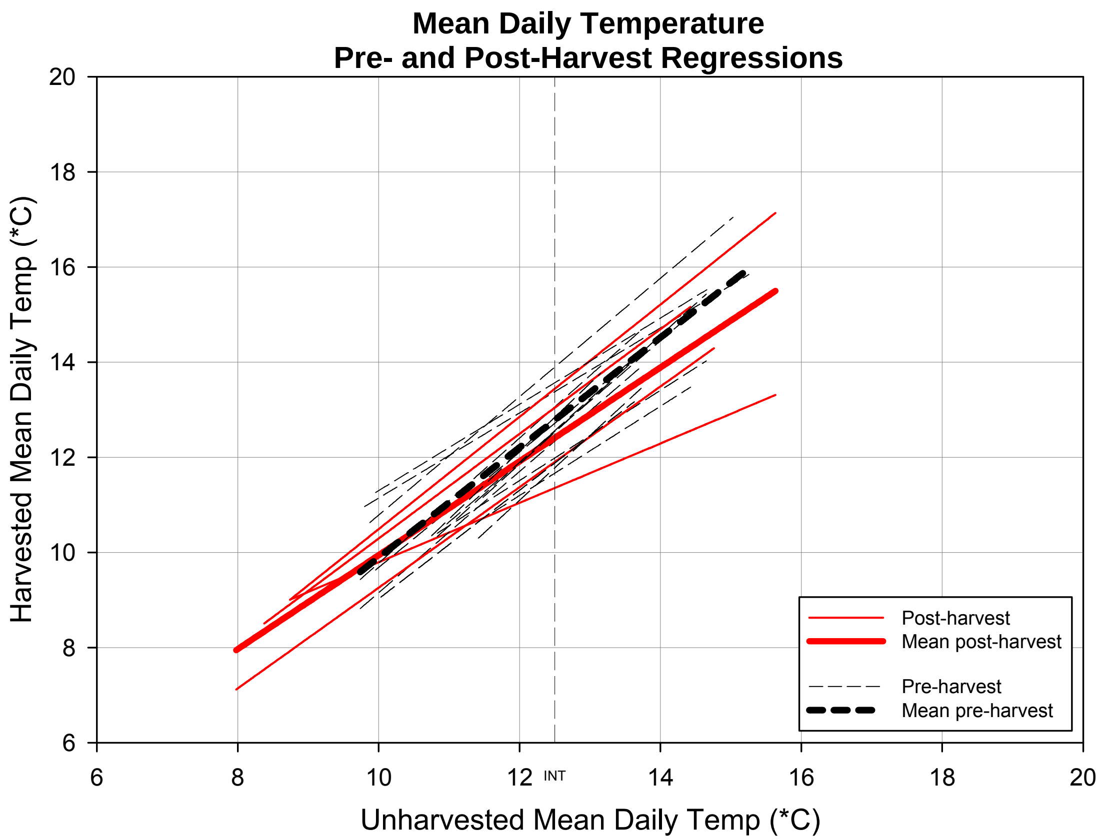


Figure 2.4c. Regressions of daily mean stream temperatures in harvested streams versus unharvested streams for each stream pair and year illustrate variability of the harvested-unharvested relationship before and after harvest. Mean pre- and post-harvest regressions illustrate comparisons made by the change detection model. Vertical dashed line indicates mean intercept.

Diel temperature fluctuation

The post-harvest ratio of harvested to unharvested diel temperature difference was found to be significantly greater than the pre-harvest ratio for every period of the summer except for the period from June 1 to June 14. The following table summarizes the differences between pre-harvest and post-harvest ratios.

Table 2.6. Mean percent change in diel temperature fluctuation after harvesting in four harvested streams. Change is significant in every period except for June 1 to June 14.

Period	Dates	Change	95% CI	DF	t-stat	p-value
1	6/1 to 6/14	49% greater	0 to 123% greater	8	2.27	0.0533
2	6/15 to 6/30	71% greater	25 to 135% greater	8	3.93	0.0043
3	7/1 to 7/14	79% greater	29 to 148% greater	8	4.08	0.0035
4	7/15 to 7/31	118% greater	63 to 193% greater	10	5.92	0.0001
5	8/1 to 8/14	137% greater	88 to 199% greater	10	8.29	<0.0001
6	8/15 to 8/31	97% greater	46 to 166% greater	10	5.05	0.0005
7	9/1 to 9/14	139% greater	96 to 190% greater	10	9.87	<0.0001
8	9/15 to 9/30	71% greater	27 to 128% greater	8	4.21	0.0030

The change between pre-harvest and post-harvest ratios can be interpreted to indicate that the diel range of stream temperatures was significantly greater after harvesting than before.

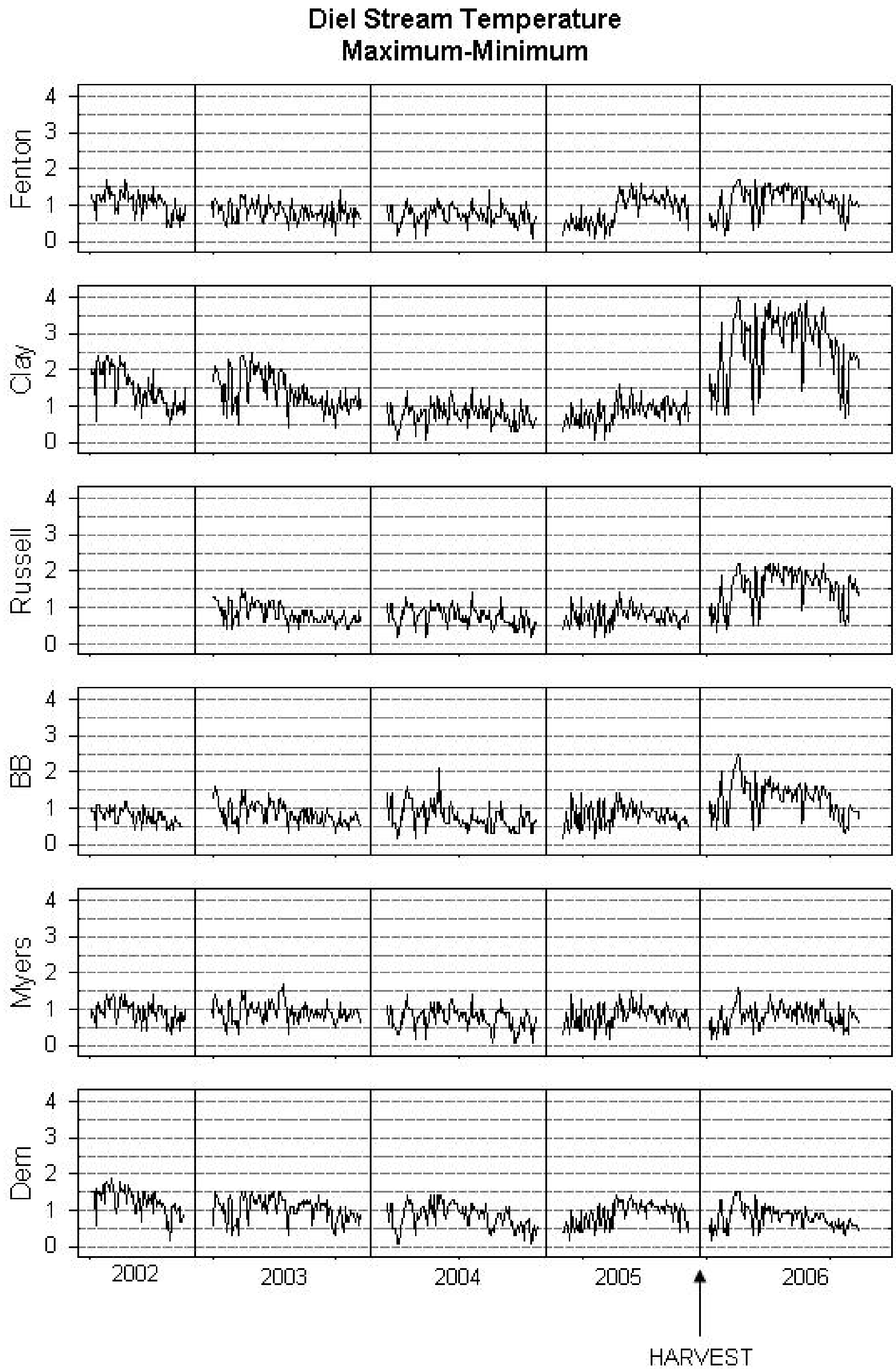


Figure 2.5. Diel fluctuation in stream temperature for every stream pre- and post-harvest. DeMerrseman and Myers are unharvested.

Greatest annual seven-day moving mean of the maximum daily temperature

Statistically significant changes to the magnitude of annual maximum seven-day moving mean of daily maximum temperatures were not detected following harvest at Hinkle Creek. The following table summarizes annual maximum seven-day mean for each stream pair and compares mean pre-treatment maximum seven-day mean to the post-treatment maximum seven-day mean.

Table 2.7. Differences between mean pre-harvest annual maximum seven-day mean stream temperatures and post-harvest annual maximums in each stream. Myers and DeMerrseman are unharvested.

Stream	Pre-treatment mean (2002-2005) ˚C	Post-treatment (2006) ˚C	Change (Post-Pre) ˚C
Fenton	14.9	13.9	-1
Clay	16.3	18.6	2.3
Russell	14.4	15.2	0.8
BB	14.6	15.7	1.1
Myers*	15	16	1
DeMerrseman*	14.2	14.8	0.6

*unharvested

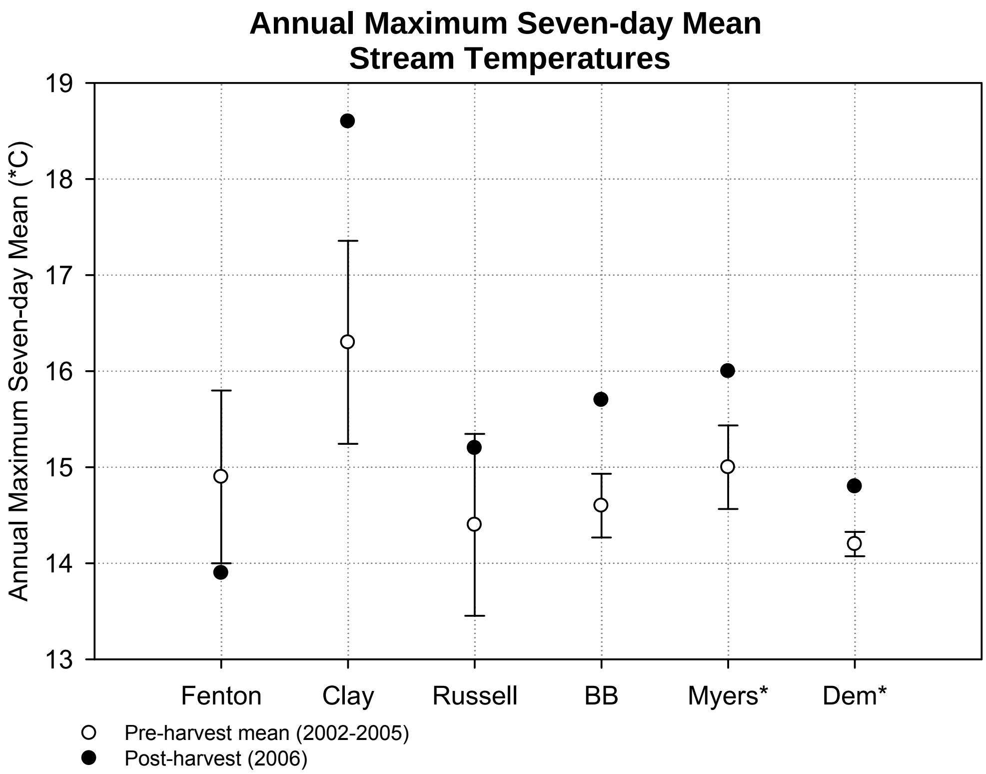


Figure 2.6. Annual maximum seven-day mean stream temperature in all streams, pre- and post-harvest. Error bars display one standard deviation from the mean of four pre-harvest years. *Myers and DeMerrseman are unharvested.

Cumulative degree days

Degree day accumulation for 2006 (post-harvest) is similar to pre-harvest years and patterns of degree day accumulation are similar between harvested and unharvested streams (Figure 2.7).

Canopy closure

A comparison of canopy closure observations taken in unharvested reaches (Figure 2.8) using a densiometer in 2004 and 2006 indicated that the 2004 densiometer crew measured 4% greater canopy closure than the 2006 crew. A similar comparison of canopy closure observations taken in 2004 and 2006 using the densiometer and the photo method revealed that the 2006 densiometer method measured 9% greater canopy closure than the photo method and the 2004 densiometer survey measured 13% greater canopy closure than the photo method. These differences are taken to represent a measure of error between the three surveys. Accounting for error between surveys allows for comparison of canopy closure measurements among the three surveys.

According to the 2004 pre-harvest densiometer survey, all reaches had greater than 95% mean canopy closure prior to harvest, with the exception of Clay DS which was harvested in 2001 before the onset of the Hinkle Creek study (Figure 2.9). The riparian zone surrounding first 100 meters of Clay DS was not harvested to provide trees for wildlife while the remainder of the reach was clearcut harvested.

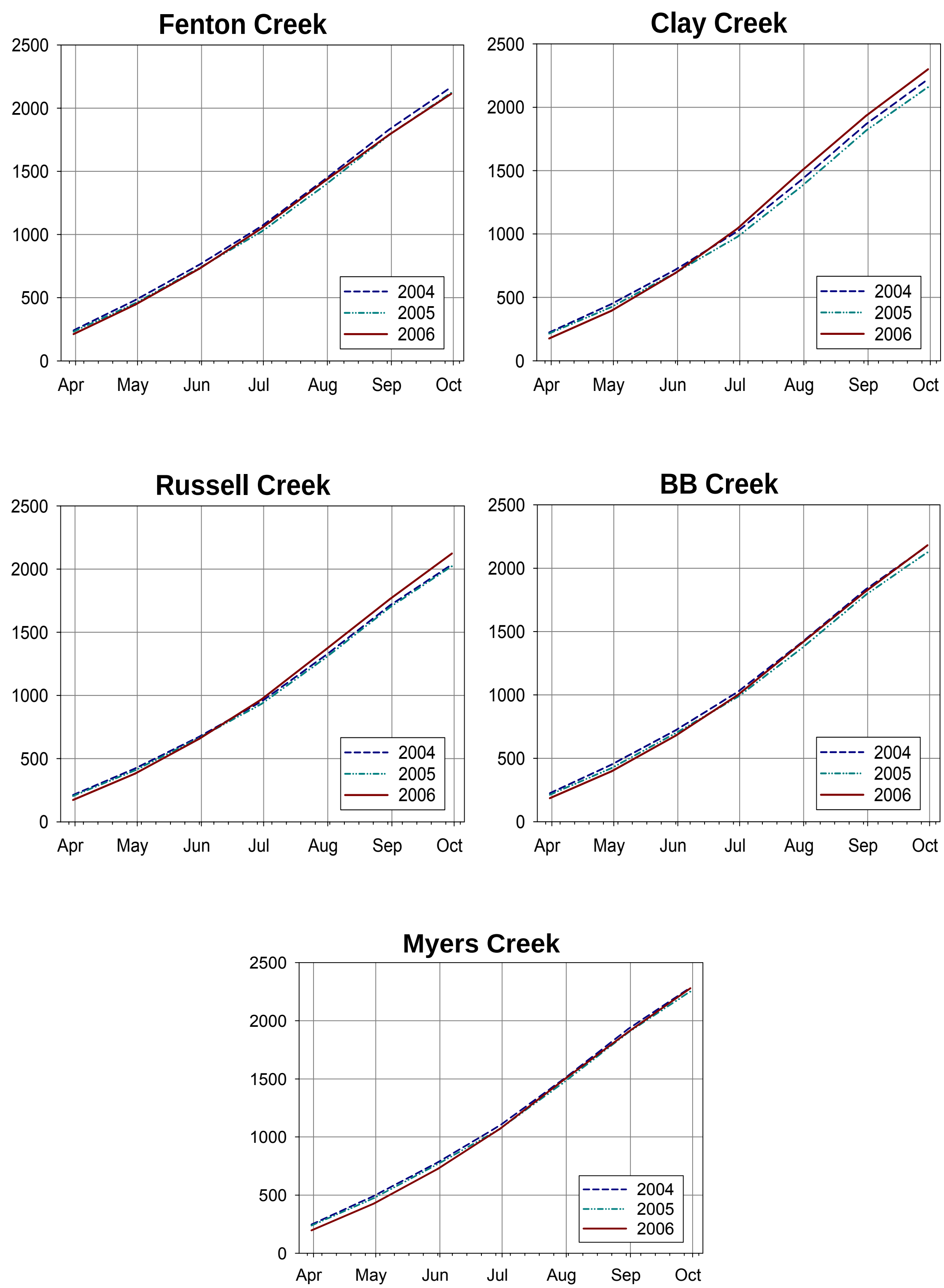


Figure 2.7 Cumulative degree days in four harvested and one unharvested stream for 2004, 2005 and 2006. Degree-day accumulation begins each year on March 1 and ends on September 30.

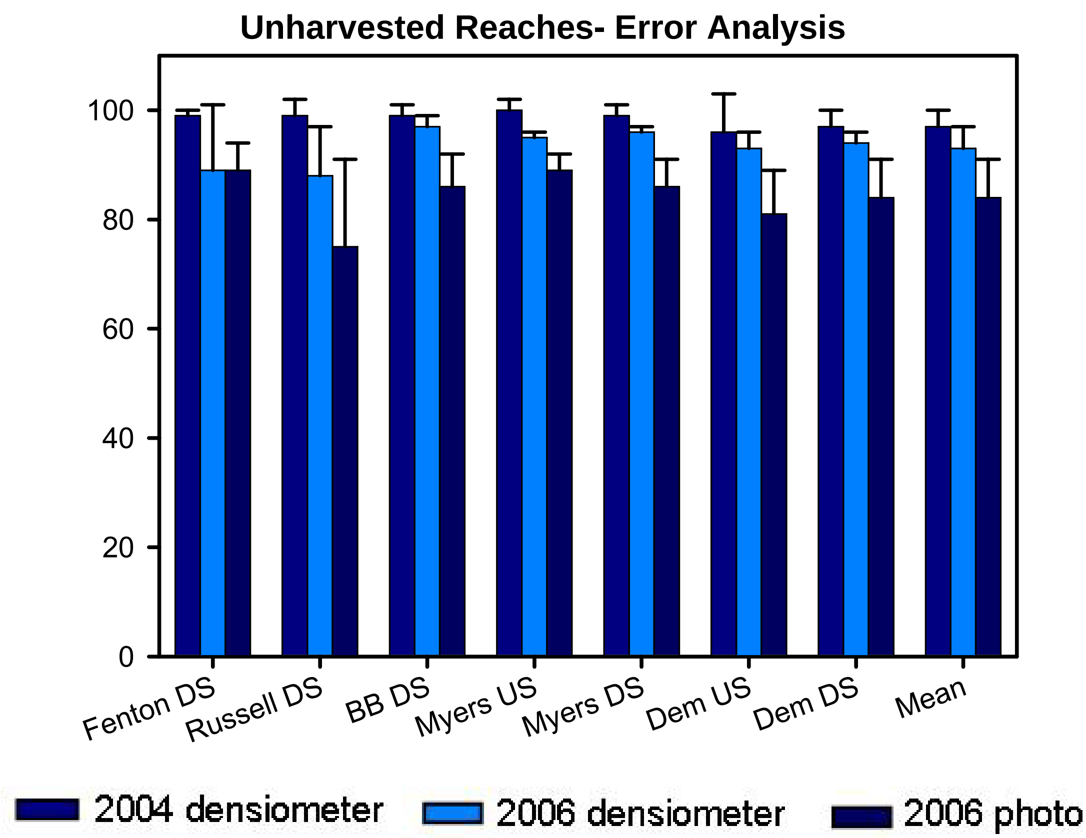


Figure 2.8. Error analysis: Percent canopy closure for all unharvested reaches. Error bars are one standard deviation of the mean. Final group represents mean values across all unharvested reaches.

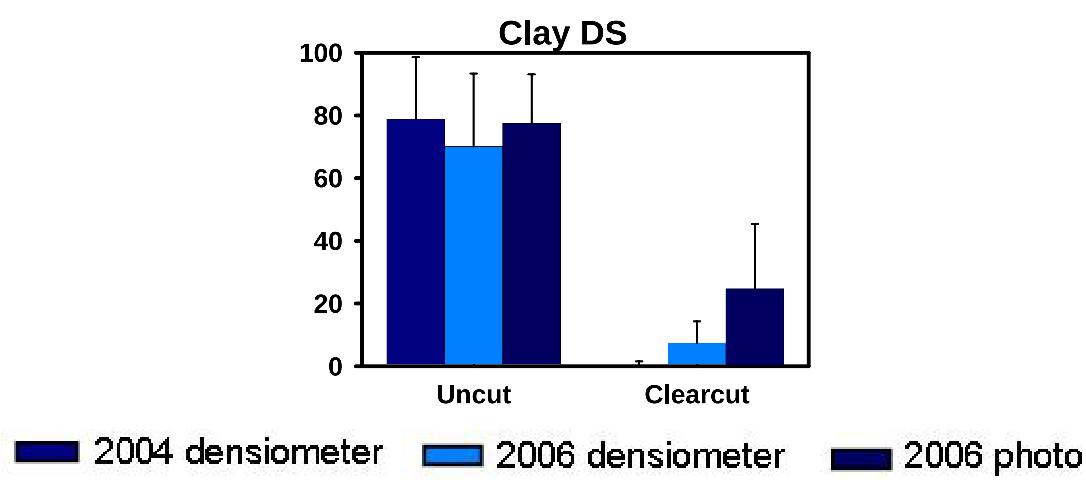


Figure 2.9. Percent canopy closure for uncut and clearcut portions of the Clay DS reach which was harvested in 2001. Error bars are one standard deviation of the mean.

The post-harvest densiometer survey indicates that canopy closure in harvested reaches decreased by 84% on average after harvesting, taking into account error between the 2004 and 2006 crews, whereas there was no change to canopy closure in unharvested reaches (Figure 2.10). However, the 2006 photo survey indicates that canopy closure decreased by 20% when error between the 2006 photo method and 2004 densiometer method is accounted for. Similarly, there was no difference in canopy closure between the densiometer method and the photo method after error between the two methods was accounted for in unharvested reaches.

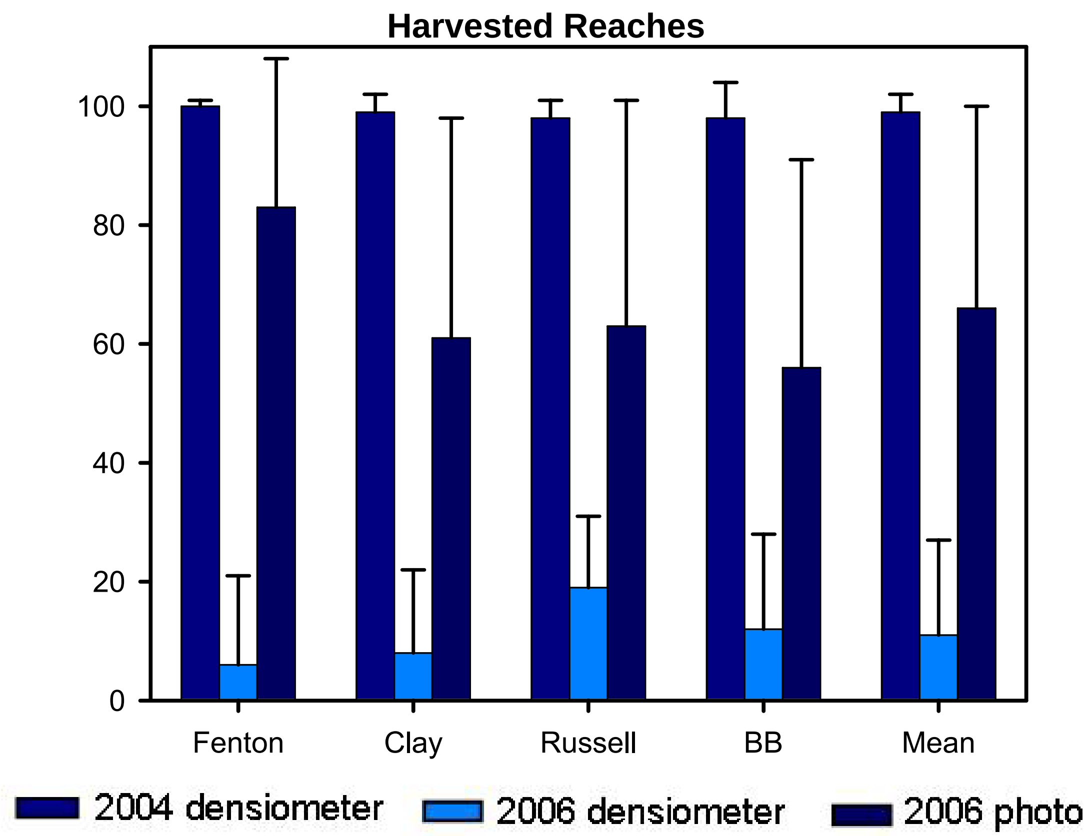


Figure 2.10. Percent canopy closure in harvested reaches. Error bars are one standard deviation of the mean. Final group represents mean values across all harvested reaches.

Table 2.8. Percent canopy closure and standard deviation in each surveyed reach before and after harvest. Fenton US, Clay US, Russell US and BB US were harvested in 2005. Clay DS was harvested in 2001.

Reach	2004 Densiometer	2006 Densiometer	2006 Photo
Fenton US*	100±1	6±15	83±25
Clay US*	99±3	8±14	61±37
Russell US*	98±3	17±19	63±38
BB US*	98±6	12±16	56±35
Myers US	100±2	95±1	89±3
DeMerrseman US	96±7	93±3	81±8
Fenton DS	99±1	89±12	89±15
Clay DS**	23±38	30±34	42±31
Russell DS	99±3	88±9	75±16
BB DS	99±2	97±2	86±6
Myers DS	99±2	96±1	86±5
DeMerrseman DS	97±3	94±2	84±7

* harvested winter 2005; **harvested 2001

Discussion

Analysis

The experimental design of Before After Control Intervention (BACI) studies intended to detect ecological change on the catchment scale, in particular paired watershed studies, is criticized due to costs associated with research on a watershed scale, pseudoreplication of experimental units and the difficulty of drawing causal inference that can be applied outside of the studied area (Hewlett 1973, Hurlbert 1984). However, using data from a paired control watershed as an explanatory variable to predict the response of a specific parameter of interest in a treated watershed can greatly increase the statistical power of change detection models when data observed in the treated and control watersheds are highly correlated (Loftis et al. 2001). The basic structure of a paired watershed investigation includes three distinct phases. During the calibration period, data are collected from paired treatment and control watersheds, which are both undisturbed and assumed to be in a state of equilibrium relative to one another with respect to the parameter of interest. Data recorded during the calibration phase establish the pre-treatment relationship between the treatment and control watersheds and characterize the inherent variability of that relationship. During the second phase, the treatment watershed is disturbed while the control watershed remains undisturbed. The third phase entails a period of post-treatment data collection from both watersheds and analysis focuses on detecting differences between the pre-treatment relationship and the post-treatment relationship. A key assumption made in all paired watershed studies is that the relationship between treated and control areas remains stable over time and that significant changes to the treatment-control relationship occur only due to the perturbation of the treated areas. Subtle fluctuation within the treatment-control relationship that occurs among pre-treatment years of data collection characterize an envelope of natural variability for the relationship and post-treatment changes to the relationship that exceed this envelope constitute significant treatment effects. Within the Hinkle Creek study, the assumption of a stable relationship between stream temperatures in harvested and unharvested streams allows for detection of a harvest effect if the relationship changes significantly following forest harvesting relative to the natural pattern of variability recorded during the calibration years.

Stream temperatures in the harvested and unharvested streams of Hinkle Creek are highly correlated (Table 2.4) thus, including the explanatory variable of stream temperature observed in the unharvested streams as a stable predictor of temperature in the harvested streams greatly enhances the power of the change detection model and reduces the probability that a Type II error will occur during analysis (Loftis et al. 2001). In order to detect changes to daily maximum, minimum and mean stream temperatures in the harvested streams, a pre-harvest relationship between each harvested and unharvested stream was defined by slope and intercept parameters of the harvested-unharvested pair regression line (Tables A1-A3, Appendix A). These regression parameters impart information about how each harvested stream responds to thermal fluxes relative to its unharvested counterpart and differences between the pre-harvest and post-harvest relationships are related through changes to these parameters. There are four possible outcomes of change between the pre-harvest and post-harvest relationships:

1. intercept could change while the slope remains stable,
2. slope could change while the intercept remains stable,
3. slope and intercept could change, or
4. slope and intercept could remain stable.

A change to the intercept parameter alone signifies that the harvested-unharvested relationship remains stable between years, but that every observation in the harvested stream is shifted up or down relative to its position in previous years (Figure 2.11).

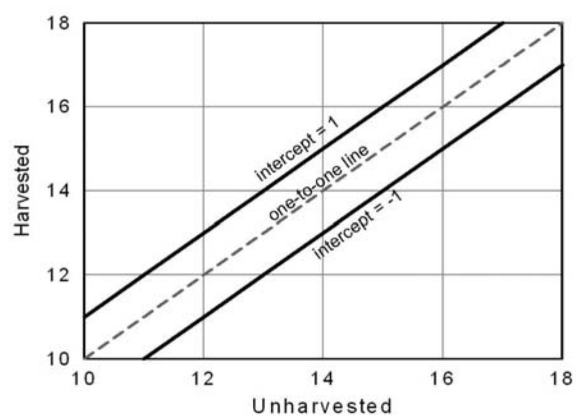


Figure 2.11. Comparison of lines with same slopes but different intercepts.

A slope greater than one indicates that for every one degree temperature increase or decrease in the unharvested stream, temperature in the harvested stream increases or decreases more than one degree (Figure 2.12). Slopes of greater than one signify more extreme temperature fluctuation in the harvested stream as compared to the unharvested stream.

Likewise, a slope of less than one indicates a damped temperature response in the harvested stream as compared to the unharvested stream.

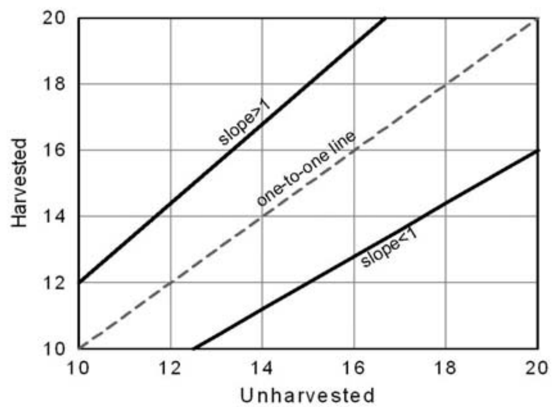


Figure 2.12. Comparison of regression lines with different slopes but same intercept. Slopes are greater than one, equal to one and less than one.

Lines that have different slopes (are not parallel) must eventually cross and if the cross occurs within the range of observed data, the conclusion of whether stream temperatures increased or decreased may vary depending on the range of temperatures in question. An increase in slope does not necessarily indicate that all stream temperatures in the range of observation increased. If the slope of the post-harvest regression increases compared to the pre-harvest slope while the intercept remains stable, this indicates that all temperatures greater than where the pre-harvest and post-harvest lines meet are greater after harvesting than before harvesting. Temperatures that fall below where the pre- and post-harvest lines cross may be cooler in the harvested stream after harvesting. If a difference between pre- and post-harvest slopes occurs in conjunction with a divergence between pre- and post-harvest intercepts, it is possible that the direction of post-harvest stream temperature response may vary even more dramatically depending upon the range of temperatures in question (Figure 2.13).

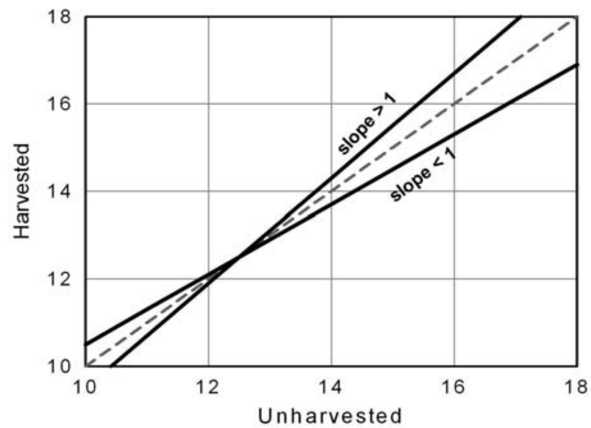


Figure 2.13. Comparison of lines with different slopes and different intercepts. Slopes are greater than one, equal to one and less than one; intercepts are -1, 0 and 1.

For example, if the slope of the post-harvest regression increased relative to the pre-harvest slope and the post-harvest intercept decreased relative to the pre-harvest intercept, it is possible that post-harvest stream temperatures could be greater than pre-harvest temperatures on the warmer end of the observed temperature range and less than pre-harvest temperatures on the cooler end. Therefore, if significant changes to either slope alone or both slope and intercept are confirmed, it is important to specify the range of temperatures over which changes occurred.

A change in slope or intercept between years in a given stream pair signifies that at least one stream is receiving or processing energy differently than in previous years. Because the unharvested watersheds remain undisturbed, it is inferred that any difference between the pre-harvest and post-harvest relationship is due to disturbance of the harvested streams. Additionally, because the pre- and post-harvest harvested-unharvested relationship are created with data from stream pairs that are geographically proximate and subjected to similar climatic conditions, the potentially confounding factor of interannual climatic variability is addressed by investigating changes to the unharvested-harvested relationship.

The significance of a change in slope or intercept after harvest depends on the magnitude of the change relative to the variability among slopes and intercepts observed during pre-harvest years. A change between the pre- and post-harvest parameter of interest must be large relative to the variance of that parameter in order to reject the null hypothesis of no change between pre- and post-harvest conditions. Regression slopes among the four pre-treatment years are stable and variability is low within individual stream pairs (Tables A1-A3 in Appendix A, Figures 2.3a-2.3c). However, variation among mean pre-harvest slopes of the four stream pairs increases variability within the change detection model, which increases the smallest difference in pre- and post-harvest slopes that can be considered statistically significant.

Intercepts vary widely among years within some individual stream pairs. A pattern of shifting intercepts between years was observed in the calibration relationships of Fenton and Clay Creeks (Figures 2.3a-2.3c). Both harvested streams were paired with Myers Creek as the unharvested stream. Data from 2003 and 2004 cluster together as do data from 2002 and 2005 and intercept values from 2003 and 2004 regressions are on the order of 1 to1.5˚C greater than intercepts from 2002 and 2005 regressions, which increases variability within the intercept parameter of these two streams. In contrast, Russell and BB Creeks were paired with DeMerrseman Creek and less interannual variability among intercept parameters exists in Russell and BB regressions than in Fenton and Clay regressions. The difference in variability between stream pair regressions is easily observed when the size of 95% prediction intervals around Fenton and Clay regressions are compared to prediction intervals around Russell and BB regressions (Figures 2.3a-2.3c). The fluctuation of intercept parameters before harvest most likely occurred because of differences in hydrologic variables between years in Fenton and Clay Creeks. This fluctuation in the intercept parameters does not invalidate the calibration relationships, but rather characterizes the variability that can be expected between undisturbed stream pairs.

Maximum, minimum and mean daily stream temperatures

The regression parameters of the post-harvest regressions of maximum daily stream temperatures were not significantly different than pre-harvest regression parameters, which indicates that maximum daily stream temperatures in harvested streams did not increase significantly after forest harvesting. These results are contrary to findings reported in several past BACI studies that examined effects of forest harvesting on temperatures of small streams in the Pacific Northwest. In similar paired watershed investigations, maximum daily stream temperatures often increased after forest canopies were removed (Levno and Rothacher 1967, Brown and Krygier 1970, Gomi et al. 2006, Macdonald et al. 2003). However, Jackson et al. (2001) reported minimal change to stream temperatures in western Washington headwater streams following clearcutting.

Slopes of post-harvest minimum and mean daily stream temperature regressions were significantly less than pre-treatment regression slopes while post-harvest intercepts were not significantly different than pre-treatment intercepts. Over the range of stream temperatures observed, the lower slopes indicate that on most days, minimum and mean daily stream temperatures decreased after harvesting at Hinkle Creek (Figures 2.4c and 2.4c). Changes to minimum stream temperatures are not as widely cited in stream temperature literature as changes to maximum temperatures, likely because the temperature standards of most States are created to address maximum temperatures. However, some research has reported significant decreases to minimum daily temperatures after forest harvesting (Johnson and Jones 2000, Macdonald et al. 2003).

Plots of 95% prediction limits around pre-harvest regression lines function not only to allow visual characterization of the variance of pre-treatment relationships, but also permit identification individual post-harvest departures from predicted values (Figures 2.3a-2.3c). By definition of the 95% prediction interval, one would expect 5% of the post-treatment data to fall outside of the prediction limits, even in lieu of a significant treatment effect.

Examination of regression plots with 95% prediction limits reveals that in 9 out of 12 plots, over 5% of post-treatment data fall outside of 95% prediction limits and that there is a consistent pattern to the departures. Whether the departures fall above the upper 95% prediction limit as in daily maximum temperatures of BB Creek or below the lower 95% prediction limit, as seen in daily maximum temperatures of Fenton Creek (Figure 2.3a), almost every departure from the 95% prediction interval is observed when temperatures in the unharvested stream are greater than 12˚C. The 12˚C threshold is consistent among daily maximum, minimum and mean temperatures. This pattern of departure from the 95% prediction interval indicates that the most significant changes between pre- and post-harvest stream temperatures occurred on days when daily maximum, minimum and mean stream temperatures exceeded 12˚C.

This is an important piece of information to consider when interpreting the slope decreases observed in minimum and mean daily temperature regressions. Lower stream temperatures were observed after harvesting in the harvested streams when the temperature in the unharvested stream was greater than 12˚C. When stream temperatures in the unharvested streams were below the 12˚C, stream temperatures in the harvested streams were similar to pre-harvest temperatures. The pre-harvest and post-harvest slopes were significantly different, and were not parallel and so the lines must cross at some temperature value in the unharvested stream. This temperature in the unharvested stream is a threshold and when minimum or mean daily temperatures are above this threshold value, minimum and mean stream temperatures in the harvested streams were lower after harvest than before harvest. The cross occurred when the minimum daily temperature in the unharvested streams was 9˚C and the mean daily temperature was 10.3˚C. In summary, minimum daily stream temperatures in harvested streams were lower after harvesting when minimum temperatures in the unharvested streams were greater than 9˚C and did not change when minimum temperatures in the unharvested streams were cooler than 9˚C. Likewise, mean daily stream temperatures in harvested streams were lower after harvesting when mean temperatures in the unharvested streams were greater than 10.3˚C and did not change when minimum temperatures in the unharvested streams were cooler than 10.3˚C.

Diel temperature fluctuation

Throughout the summer, diel stream temperature range fluctuates in a pattern of higher diel range during the mid-summer weeks and lower fluctuation at the beginning and end of the warm season (Figure 2.5). As such, it is unreasonable to compare diel stream temperature fluctuations from the beginning or end of the warm season to temperature ranges that occur during the mid-summer weeks. In order to avoid such unrealistic comparisons, the warm season (June 1 to September 30) was partitioned into eight discrete periods that were analyzed separately.

The highly significant differences observed between pre- and post-harvest diel stream temperature fluctuations at Hinkle Creek are similar to results reported for other comparable stream temperature studies (Brown and Krygier 1970, Johnson and Jones 2000). Johnson and Jones [2000] observed that diel range in harvested streams was much greater than in unharvested streams and that diel fluctuation in the harvested streams recovered to magnitudes comparable to unharvested streams after the riparian canopy recovered to pre-harvest levels. Brown and Krygier [1970] reported that diel temperature fluctuations increased dramatically in a clearcut watershed whereas diel fluctuations in an undisturbed and patch-cut watershed did not change appreciably. Most studies that cite differences between pre-harvest and post-harvest diel stream temperature fluctuations often also report significantly greater maximum daily stream temperatures, which were not observed at Hinkle Creek. Rather, the significantly lower minimum daily stream temperatures observed at Hinkle Creek was likely the source of the wider diel fluctuations observed after harvesting.

Diel stream temperature ranges recorded in 2005 in Fenton Creek illustrate the nearly immediate effect of forest harvesting on diel stream temperature fluctuations (Figure 2.5). Fenton Creek was the first harvest unit to be felled and was cut during the summer of 2005. Data from 2005 in Fenton Creek were removed from analysis because data from half of this summer reflect clearcut conditions. On July 14, 2005, diel stream temperature fluctuation nearly doubles as compared to ranges observed the week prior. This date coincides closely with the start of harvesting in Hinkle Creek.

Degree days

Plots of cumulative degree days for harvested streams beginning on March 1 indicate little change in degree day accumulation between pre-harvest years and the post-harvest year (Figure 2.7). Analyses of mean daily temperatures during the warm season (June 1 to September 30) indicate that mean daily stream temperatures decreased in every stream. The decrease in warm season mean daily temperature was not apparent in degree day accumulation starting on March 1 as three of the four harvested streams exceeded pre-harvest degree day accumulation by early July 2006. By October 1 in 2006 Clay Creek had accumulated 78 (3%) more degree days than in 2004 and 140 (6%) more degree days than 2005, Russell Creek had accumulated 86 (4%) more days than 2004 and 100 (5%) more than 2005 and BB Creek had accumulated 4 (0.2%) more days than 2004 and 54 (2.5%) more days than 2005. Cooler mean temperatures were apparent in Fenton Creek which accumulated 53 (2.5%) less degree days in 2006 than in 2004 and 8 (0.4%) days less than 2005. The cumulative degree day plot for Myers Creek (unharvested) demonstrates that 2006 was similar to 2004 and 2005 in terms of degree day accumulation in an undisturbed stream. Johnson and Jones [2000] reported that degree days accumulated more rapidly in an unshaded clearcut stream and a stream scoured by a debris flow than in shaded streams but also reported increases to mean maximum and minimum weekly temperatures in the unshaded streams.

Experimental design and individual stream reach analysis

Pseudoreplication is a common criticism of past paired watershed study designs as many seminal paired watershed studies have based their conclusions on the response of single iterations of applied treatments and employed statistical methods that were designed for replicated studies (Hurlbert 1984). The Hinkle Creek stream temperature study is a paired watershed experiment where the harvesting treatment was applied to multiple experimental units. Within the experimental design of the Hinkle Creek study, the four harvested streams represent four replicates of the harvesting treatment and the average response across the four streams constitutes the overall response. While the replicated experiment is necessary to allow for correct application of hypothesis testing, it is also informative to scrutinize the response of each individual stream. Examination of stream temperature responses and variables that may influence stream temperature at the individual reach level may allow for more comprehensive conclusions to be drawn pertaining to processes that influence stream temperature patterns.

Significant changes to maximum daily stream temperatures were not detected at Hinkle Creek when the mean response of all four harvested streams was considered. An overall response of no change to the unharvested-harvested relationship after harvesting may imply that no change was observed in any of the four individual relationships, which is misleading. When the four streams are considered individually, it is evident that slopes of daily maximum temperature regressions changed significantly in Fenton and BB Creeks after harvest. The post-harvest slope in Fenton Creek was 0.28 (30%) lower than the mean of the pre-harvest slopes and the post-harvest slope in BB Creek was 0.30 (37%) higher than the mean of the pre-harvest slopes (Table 2.5a). There was no appreciable change to post-harvest slopes in Clay and Russell Creeks and as the response vectors from Fenton and BB Creeks were approximately equal in magnitude and opposite in direction, the net change became zero (Figure 2.3a, Table 2.5a).

A similar pattern emerges when the four streams are considered individually in the analysis of annual maximum seven-day mean. Once again, it is helpful to consider the annual maximum temperatures observed in the unharvested streams as a prediction of the annual maximum temperatures that should occur in the harvested streams if there were no change. Annual maximum seven-day mean stream temperatures in the two unharvested streams were 0.5 to 1˚C greater in 2006 than the average of the four pre-harvest years (Table 2.7, Figure 2.6). A similar pattern in the harvested streams should be observed if there were no changes to stream temperatures due to harvesting. The difference between post-harvest and pre-harvest annual seven-day maximums in Russell and BB Creeks was comparable to the difference observed in the unharvested streams, however the annual maximum was 1˚C lower than the average in Fenton Creek and 2.3˚C higher than the average in Clay Creek. Once again, although changes to annual maximum seven-day mean were observed in individual streams, because the streams responded divergently, the overall result is no net change. The pattern of divergent response among the four harvested streams was not observed in minimum and mean daily stream temperature relationships. Slopes of the unharvested-harvested regressions of minimum and mean daily stream temperature decreased after harvesting in all four streams.

Divergent responses among experimental replicates suggest that the effect of treatment was not great enough to stand out beyond the natural variability of the studied experimental units. However, when systems as complex as streams are investigated, one must question whether the temporal and spatial heterogeneity inherent to stream reaches renders the individual stream undesirable as an experimental replicate. The replicated experimental design was developed to detect changes to one isolated variable while all other variables are held constant. The assumed consistency of other factors implies that some level of control must exist over the remaining variables. This level of control is nearly impossible to achieve when working with natural systems, particularly with replicates that are as variable and complex as streams. Heterogeneity in microclimatic factors, surface discharge patterns, stream morphology, and delivery and exchange of water through changing subsurface flowpaths may affect stream temperature patterns from year to year and from stream to stream within a given year. If each variable that could potentially influence stream temperature were controlled, experimental units of replicated concrete troughs would replace actual streams in order to isolate the one variable of interest. However, from a management perspective, such a controlled experiment would not provide the desired information about the effects of forest harvesting on natural streams. Therefore, the inherent variability of streams as replicates must be addressed in any experiment designed to detect stream temperature changes. The use of data from an unharvested stream addresses interannual variability of landscape-scale factors such as climatic variability, but we are still left with many complex processes and interactions within the entity of the individual stream that may be different in the treatment stream and the paired control or between harvested replicates. Investigating changes observed on the level of the individual stream reach rather than on the scale of a replicated experiment can help to identify some of the processes that lead to the observed responses. Additionally, reach-level documentation of variables known to be important to the process of stream heating can be used to explain changes that we observe in each individual stream and perhaps to construct a conceptual framework of the dominant processes that led to the observed stream temperature patterns.

Canopy closure

Based upon results from similar temperature studies in headwater streams in the Pacific Northwest and on the principles of thermal dynamics for a small stream discussed in Brown’s energy balance, the primary *a priori* hypotheses for the Hinkle Creek stream temperature study were that maximum daily stream temperatures would increase significantly, minimum daily temperatures would decrease slightly and mean daily temperatures would increase slightly or remain stable after harvesting. After documenting different results than were hypothesized, it is evident that the suite of processes that control reach scale stream temperatures are not fully understood at this point, or that more specific information is needed to explicate the results. One important piece of information that may partially account for the observed temperature response is the change in solar radiation exposure between pre-harvest and post-harvest years. Absorption of solar radiation is the primary mechanism that causes stream temperatures to increase (Brown 1969, Beschta et al. 1987, Johnson and Jones 2000, Johnson 2004) and as the level of shade over a stream is a significant control to the amount of solar radiation that reaches the stream surface, shade is a crucial determinant of stream temperature patterns (Brown and Krygier 1970, Levno and Rothacher 1967). Although an intact forest canopy is the traditional and most widespread mechanism of stream shading, researchers have demonstrated that any material that attenuates solar radiation before it reaches the stream can prevent increases to stream temperature in similar fashion to a forest canopy (Johnson 2004, Jackson et al. 2001). The anticipated results of the stream temperature study were hypothesized assuming that shade over the streams would decrease considerably after the overstory canopy was removed, leaving the streams exposed to significantly greater amounts of solar radiation. Because solar radiation is the primary driver of stream temperature, it is desirable to compare levels of solar radiation that reached the streams before and after harvesting as it is plausible that the streams did not receive the expected increase in delivery of solar radiation.

Often in forestry and ecological research, rather than taking direct measurements of solar radiation, which is costly and time-consuming, researchers quantify levels of canopy openness to use as a proxy for available solar radiation. Jennings et al. [1999] defines canopy openness as the proportion of sky that is not covered by vegetation and where solar radiation is available to reach the stream without attenuation. Canopy closure is the analog of canopy openness and represents the proportion of sky where shortwave solar radiation is attenuated before it can reach the stream and is related to canopy openness by the following equation:

$$\text{Canopy closure} = 1 - \text{Canopy openness}$$

Canopy closure was measured before and after harvesting with a hand-held spherical densiometer. The spherical densiometer was chosen because it is inexpensive, does not require extensive technical training to employ and measures canopy closure quickly. In total, 688 canopy densiometer measurements characterized twelve stream reaches in the 2004 (pre-harvest) survey and 585 densiometer measurements were taken in the 2006 (post-harvest) survey. This density of canopy closure sampling could not have been feasibly achieved using a more time-consuming method, such as hemispherical photography.

Mean canopy closure within the harvested reaches of Hinkle Creek was over 95% in every reach surveyed with a densiometer before harvesting occurred and harvested reaches had a mean canopy closure of 99%. Therefore the pre-harvest maximum daily temperatures recorded at Hinkle Creek occurred in response to less than 5% of the total available solar radiation. Daily energy balances at Hinkle Creek before harvest most likely looked similar to Brown's energy budget for a forested stream (Figure 1.1a) where evaporation, convective heat exchange and longwave radiation were comparable to incoming solar radiation. According to the survey of post-harvest canopy closure sampled with a densiometer, mean post-harvest canopy closure in the harvested reaches was 11%, meaning that the harvesting treatment reduced overstory canopy closure by 88%. An energy budget for a stream with 11% canopy closure would look more like Brown's energy budget for an unshaded stream (Figure 1.1b) where the magnitude of the incoming solar radiation term is two orders of magnitude larger than the magnitudes of sensible and latent heat flux. If the harvested streams had been exposed to 88% more solar radiation the summer after harvest than in previous years, Brown's energy budget predicts that dramatic increases in stream temperature would be observed. However, the post-harvest stream temperature data clearly indicate that stream temperatures did not increase dramatically following harvest and in fact, stream temperatures decreased in one harvested stream. Clearly, the canopy closure values obtained from the post-harvest densiometer survey underestimated the amount of shade available within the harvested reaches. A possible explanation for the underestimation is that a densiometer is read at waist height, thus cover located below waist height was not accounted for in the densiometer survey. The densiometer survey was an effective method to measure overstory canopy closure but did not provide a true approximation of solar radiation exposure in the harvested streams.

After harvesting, the harvested streams were partially covered by a layer of organic material that was left when the merchantable timber was removed. This layer of logging slash attenuated significant amounts of solar radiation before it could reach the streams. In order to estimate the true increases to solar radiation exposure that occurred as a result of the harvesting treatment, pre-harvest canopy closure and the post-harvest canopy closure that accounts for both overstory vegetation and slash cover must be compared. To quantify canopy closure that included the slash, canopy closure was measured from a perspective of just inches above the stream surface and below the intact slash layer. It was also desirable that a sampling density comparable to the sampling density measured with the densiometer survey was maintained during the slash-closure survey. An additional constraint to the method of measuring slash-closure was that the sampling device had to be small as the space between the stream and the slash layer was often tight. A 35 millimeter digital photo survey was preferred over hemispherical photography because the time constraints associated with hemispherical photography would not allow the desired sampling density and because the hemispherical equipment set-up was too large to fit underneath the slash. Therefore, during the 2006 canopy closure survey, canopy closure was sampled at each survey point with both the densiometer and a digital photo.

Comparing measurements of canopy closure obtained using the two different sampling methods is difficult, however if the error between the two methods can be quantified, the two methods can be compared directly. Seven stream reaches that did not receive a harvesting treatment were surveyed before and after harvesting. These seven reaches had an intact canopy throughout the study period and it is reasonable to assume that change to the true level of canopy closure in these reaches throughout the period of study was negligible. A comparison of canopy closure measurements in these seven reaches taken pre-harvest and post-harvest using the densiometer and photo methods reveals that the differences between canopy closure levels reported in the 2004 and 2006 densiometer surveys and the 2006 densiometer and 2006 photo surveys are consistent between stream reaches (Figure 2.8). On average, the 2004 densiometer survey shows 4% more canopy closure than the 2006 densiometer survey and the 2006 densiometer survey reported 9% more canopy closure than the 2006 photo survey. This brings the total mean error between the 2004 densiometer and 2006 photo survey to 13%. When the 13% error is taken into account, it is possible to compare pre-treatment canopy closure to post-treatment cover from overstory vegetation and logging slash. This comparison allows the reductions in cover due to the harvesting treatment to be quantified.

When the 4% error between the 2004 and 2006 densiometer surveys is considered, the harvesting treatment resulted in an 84% reduction in overstory canopy closure in harvested streams. When cover from logging slash is included in the cover estimates and error between the 2004 densiometer survey and 2006 photo survey is taken into account, canopy closure in harvested streams dropped from a pre-harvest mean of 87% to a post-harvest mean of 67%. A 20% decrease in canopy closure would result in much less dramatic increases to stream temperature than the 84% reduction that was quantified by the densiometer survey.

The 4% error calculated between the 2004 and 2006 densiometer surveys can be attributed to operator error. Two different field crews collected data during the 2004 and 2006 surveys and all error between the two surveys is due to the different operators. The 9% difference between the 2006 densiometer and photo surveys is due to the fact that the two methods sample different areas of the canopy. A spherical densiometer samples approximately an 180˚ view whereas the area of canopy sampled by the 35 millimeter camera lens is smaller. The wider angle of the densiometer accounts for cover that attenuates solar radiation all solar angles throughout the day whereas the photo mainly samples cover that attenuates light during peak solar angles. The different sampling area is probably the main reason for the 13% difference in canopy closure estimated by the two methods.

Past research that examined the effect of forest harvesting on stream temperatures of small streams has often reported that maximum stream temperatures increased dramatically following harvesting (Levno and Rothacher 1967, Levno and Rothacher 1969, Brown and Krygier 1970, Gomi et al. 2006). Most of the sizable increases observed occurred when all logging slash was removed from the stream. Maximum stream temperatures in Watershed 1 of the HJ Andrews Experimental Forest were 2˚C higher than predicted values after logging but were 7.5˚C higher than predicted after logging slash was removed from the stream and burned (Levno and Rothacher 1967, Levno and Rothacher 1969). Likewise, maximum stream temperatures did not increase when Watershed 3 of the HJ Andrews was patch-cut with buffers, however when debris flows scoured the channel and removed the riparian vegetation and downed vegetation in the stream channel, significant increases to maximum stream temperatures were observed (Levno and Rothacher 1967). Stream temperatures observed in a clearcut watershed in the Alsea Watershed Study increased by 8˚C the summer after harvesting, however greater increases were observed during the second summer after harvesting when logging slash was removed from the stream and burned (Brown and Krygier 1970). Logging slash was not removed from four streams that were clearcut without buffers in British Columbia and the maximum temperature increases in these streams varied between 2 and 8˚C (Gomi et al.

2006). Although logging slash was not removed from the streams, Gomi et al. [2006] state that the slash did not cover the streams or provide significant shade. The amount of shade provided by slash was not measured in the British Columbia study and it is possible that the variable maximum temperature response could be partially attributed to variable levels of shading by slash among the four streams. Finally, Jackson et al [2001] observed that maximum stream temperatures did not increase appreciably in streams that were clearcut with no buffers and covered by logging slash. The amalgamation of evidence in these studies indicates that logging slash can provide significant shade to streams and may moderate large increases to maximum stream temperatures. The absence of a significant maximum stream temperature response observed in the headwaters of Hinkle Creek can be attributed, in part, to the extensive cover provided by logging slash.

Further explanation of results

The primary physical mechanisms that dissipate heat from streams are evaporative heat flux and emission of longwave radiation (Boyd and Kaspar 2003). As evaporative flux is controlled by wind speed and vapor pressure gradients at the stream-air interface (Dingman 2002), most energy removed from the stream via evaporative heat flux is removed during the day during peak wind speeds and when the greatest vapor pressure deficit exists (Gauger and Skaugset, unpublished data). Brosofske et al. [1997] reported that forest harvesting disrupted pre-harvest riparian microclimatic gradients and that relative humidity near the stream was lower post-harvest as compared to pre-harvest values. As the vapor pressure of air is directly proportional to relative humidity, a decrease in relative humidity above the stream could lead to increased heat loss from the stream through evaporation and result in cooler minimum temperatures than would be observed under an intact forest canopy. The decreases in near-stream relative humidity observed by Brosofske et al. [1997] were not observed in clearcut conditions but rather represent conditions within buffered stream reaches. Brosofske et al. [1997] observed an exponential decrease in near-stream relative humidity as buffer width decreased thus, relative humidity could potentially be lower in clearcut streams than in the streams investigated in this study.

Brown's daily energy budget for a small stream (Figure 1.1a-1.1b) indicates that net energy fluxes directed away from the stream (negative fluxes) occur during the night (Brown 1983). Emission of longwave radiation is generally the dominant mechanism that removes heat from the stream at night (Brown 1969, Gauger and Skaugset 2004). Macdonald et al. [2003] proposed that stream temperatures were lower than expected following forest harvesting because removal of the riparian canopy allowed net heat losses through longwave back radiation to increase. It is uncertain as to whether the slash layer that covered the streams of Hinkle Creek affected longwave radiation in the same manner as an intact riparian canopy.

Although changes to the riparian microclimate and nighttime longwave radiation emission may partially explain the observed cooler minimum daily stream temperatures, and the minimal response of daily maximum stream temperatures may be partially explained by high levels of slash cover, there is also a hydrologic factor that has likely influenced the post-harvest stream temperature response. There is thorough documentation within the hydrologic literature that stream discharge increases after forest harvesting and that the effect of harvesting on streamflow varies seasonally in western coniferous forests (Harr et al. 1979, Jones and Post 2004, Keppler and Ziemer 1990, Hicks et al. 1991). In the Pacific Northwest, the largest absolute pre- to post-harvest differences in streamflow occur in the winter while greatest changes to relative streamflow occur during dry summer months (Jones and Post 2004). Harr et al. [1979] reported that summer baseflows in southwestern Oregon increased by 196% after a watershed was clearcut. Hicks et al. [1991] reported a 159% increase in late summer streamflow after logging in the HJ Andrews Experimental Forest. A significant increase in summer baseflow increases the volume of water present in the stream channel at any given time and a stream that contains a greater volume of stream water will not warm as much as a stream with a lesser volume of water. The observed increases to streamflow after forest harvesting are attributed primarily to increased inputs from subsurface sources, which have a lower temperature than the minimum daily temperatures observed during the warm season in surface waters of Hinkle Creek. Increases to summer baseflows may partially account for the lack of significant increases to maximum daily temperatures and the significant decreases to mean and minimum daily temperatures in Hinkle Creek. Increases to baseflow volume may also explain the divergent temperature responses observed in maximum daily temperatures. Changes in streamflows were documented to be related significantly to the percentage of total watershed area logged in Caspar Creek (Keppeler and Ziemer 1990). Out of the four stream replicates, the greatest percentage of the watershed was harvested from Fenton Creek (75%) and maximum daily stream temperatures decreased in Fenton Creek after harvesting (Table 2.1, Table 2.5a, Figure 2.3a), perhaps due to increased streamflow. In comparison, only 32% of the BB Creek watershed was harvested and maximum daily temperatures increased in BB Creek after harvesting (Table 2.1, Table 2.5a, Figure 2.3a).

There is an interesting opportunity to further explore the hypothesis that stream temperatures in Fenton Creek decreased after harvesting due to greater inputs of cooler subsurface water. During the summer of 2005, 75% of the Fenton Creek watershed was felled and diel stream temperature fluctuations in Fenton Creek increased immediately after the onset of felling (Figure 2.5). Diel stream temperature fluctuations increased in other streams at this time due to natural seasonal patterns in diel stream temperature, however the increases observed in Fenton Creek were abrupt and of a greater magnitude than increases observed in unharvested streams. The rapid and sizable response indicates that stream temperatures in Fenton Creek responded to felling almost immediately. Because there is often a lag time associated with streamflow increases following vegetation removal, the immediate response in Fenton Creek suggests that increased streamflow was perhaps not the cause of immediate change in diel temperature fluctuations, but that a more instantaneous factor, such as increased solar radiation, was the cause of the abrupt increase in diel fluctuations. If solar radiation were the cause of the instantaneous upsurge in diel stream temperature range, it would be evidenced by increases in maximum daily temperatures. Time-series plots of daily minimum and maximum stream temperatures in Fenton Creek and Myers Creeks (unharvested) during the summer of 2005 indicate that maximum temperatures do increase in Fenton Creek around the time of the abrupt change in diel temperature fluctuation, but that the change is similar in timing and slightly lower in magnitude as compared to changes that occur in Myers Creek at the same time (Figure A7). However, minimum temperatures in Fenton Creek appear to be lower than minimum temperatures in Myers Creek. Therefore it seems that increases in diel fluctuation are greater at Fenton Creek than in the unharvested stream due to lower minimum temperatures rather than warmer maximum temperatures. Changes to summer baseflows in Hinkle Creek were not explored in this study, however a full comparison of pre- and post-harvest summer streamflow should be completed to assess the extent to which stream temperature patterns were influenced by changes to baseflow.

Future considerations for stream temperatures in Hinkle Creek

Although the accumulation of logging slash excluded solar radiation and prevented dramatic stream temperature increases the first summer after harvesting, the thermal buffer provided by the slash is temporary. The slash is comprised of organic material that, in time, will decompose, be consumed or may be moved out of the stream or downstream by high flows. It is inevitable that over time the slash will disappear, leaving the stream increasingly more exposed to solar radiation. The rate of riparian vegetation recovery relative to the rate of slash decomposition will determine the solar radiation loading to the streams over time. In an analysis of cumulative effects of harvesting of stream temperature Beschta and Taylor [1988] assume that the effects of canopy removal on temperatures of small streams are greatest for 5 years after harvesting and that the effects decrease linearly over a period of the following 15 years until pre-harvest canopy closure levels are obtained 20 years after harvest. Similarly, Johnson and Jones [2000] observed that stream temperatures in harvested streams of the HJ Andrews paired watershed study recovered to pre-harvest conditions after full canopy closure was achieved 15 years after harvest. Similar rates of recovery may be observed in watersheds that are permitted to naturally regenerate after harvesting, however, the continued management of intensively managed watersheds may result in a trajectory of growth different from that cited by previous research. If the slash decomposes at a rate faster than the riparian vegetation grows, it is likely that the stream will be exposed to direct solar radiation and that stream temperatures will increase.

The clearcut portion of the Clay DS reach affords a convenient on-site glimpse into what canopy closure levels in the harvested reaches may resemble in five years. The Clay DS reach was harvested by Roseburg Forest Products in 2001 using similar equipment and techniques to what were used in the harvesting treatment of the Hinkle Creek study. This reach of Clay Creek is also designated as small and non-fish-bearing, thus according to the Oregon Forest Practice Rules, a RMA of merchantable timber was not left when the Clay DS reach was harvested. The 2006 photo canopy closure survey of the 2001 harvested Clay DS reach reveals that mean canopy closure from both overstory vegetation and remaining downed vegetation five years after harvest was 25%. Similar site preparation and herbicide treatments were used in the 2001 Clay DS harvest and the 2005 harvest. Therefore, it is reasonable to assume that the levels of canopy closure from overstory vegetation and slash observed in the Clay DS reach in 2006 will be similar to the levels of closure expected in the 2005 harvested streams in five years. Current plans for the future of the Hinkle Creek study include continued monitoring of stream temperatures in the 2005 harvested reaches and it is possible that this prediction can be tested in the future.

Another variable that may influence stream temperature patterns in the future is the recovery of summer baseflows to pre-harvest levels. Streamflow data from watersheds in western Oregon and California that were harvested and regenerated indicate that summer low flows increase for the first ten years following harvest, most likely as a result of reduced evapotranspiration, but as the forest matures, summer streamflow decreases relative to pre-harvest levels (Keppeler and Ziemer 1990, Hicks et al. 1991, Jones and Post 2004). The methods of site preparation following logging varied among sites that contributed streamflow data and range from broadcast burning and natural regeneration to replanting and herbicide application. Site preparation methods that restrict vegetation growth, such as herbicide treatment, are likely to hinder baseflow recovery whereas methods such as broadcast burning and natural regeneration can be expected to expedite baseflow recovery by promoting vegetation growth. The harvest units of Hinkle Creek were not burned and site preparation included multiple herbicide applications, so it is probable that baseflow will recover slowly at Hinkle Creek. The future stream temperatures in harvested reaches of Hinkle Creek will depend on the relative rates of streamflow recovery, riparian vegetation regrowth and slash decomposition.

In addition to the fact that the logging slash is only a temporary mechanism to exclude solar radiation, there are ecological problems that may arise from the input of such large quantities of organic matter into the stream system. As the slash decomposes, the biological oxygen demand (BOD) within the stream will increase and dissolved oxygen (DO) concentrations will be depleted (Berry 1975, Moring and Lantz 1975). The streams investigated at Hinkle Creek are high-gradient and the water likely reaerates quickly following DO depletion (Ice and Brown 1978); however, DO concentrations in lower gradient streams may be negatively affected. Accumulated slash disrupted riffle sequences in a clearcut stream in the Alsea Watershed study which decreased reaeration rates and exacerbated low DO concentrations (Lantz 1971). Additionally, large inputs of logging slash can alter channel morphology and particle size distribution (Jackson et al. 2001) which can potentially affect habitat quality for aquatic biota. Streambed gravels that are clogged with fine particles are not suitable habitat for salmonid spawning and so a reduction in particle sizes brought about by slash accumulation in streams may impair salmonid habitat. The Oregon Forest Practice Rules address logging slash accumulation in order to minimize impacts to water quality and prevent mass debris movement. Operators are instructed to fell away from streams, use logging practices that reduce slash movement on steep slopes and are required to remove slash that may enter streams that support fish or domestic water use within 24 hours. The Rules regarding logging slash are less specific for streams that do not support fish or domestic water use where operators are simply instructed to minimize slash accumulation but are not required to physically remove slash from the stream (ORS 629-630-0600).

Hindsight

If I were to redo this study, I would ensure that the temperature probes were deployed each year early in the growing season. In years 2004, 2005 and 2006 stream temperatures were recorded with Campbell Scientific data loggers that remained in the stream year-round and were located within feet of the HOBO data loggers that supplied primary data. Data from the Campbell Scientific loggers were used to fill in data gaps in the early part of the seasons 2004, 2005 and 2006. I also would have encased the probes in white PVC solar shields every year rather than only the post-harvest year. Data from one location in 2002 was not used because direct absorption of solar radiation corrupted the data. I also would have requested that the harvesting treatment begin after September 30 so that data from all streams taken during the summer of 2005 could be used. Finally, I would have sampled the harvested streams for DO concentration pre- and post-harvest to see if there was an appreciable difference in DO concentrations due to the large input of organic matter. Although pre- and post-harvest comparisons of DO concentration were not undertaken in this study, concurrent investigations into aquatic invertebrate and amphibian populations should document any degradation of aquatic habitat that occurs as a result of harvesting. A thorough investigation into changes to summer baseflows must also be undertaken in order to present a complete picture of the conditions under which these stream temperature results occurred.

Chapter III: Conclusions

Conclusions

Summer stream temperatures were monitored for five years in six headwater streams of the Hinkle Creek basin in southern Oregon. Between the fourth and fifth summer, a harvesting treatment was applied to four of the streams while the other two streams remained undisturbed. Harvest units were logged according to current Oregon Forest Practice Rules and modern harvesting technology was employed. Because the four harvested streams were designated as small and non-fish-bearing, a vegetated riparian buffer was not left between the streams and the harvest units. The harvesting treatment was intended to represent conditions present in intensively managed, privately owned forest land. As the Hinkle Creek basin is situated on forest land owned and intensively managed by Roseburg Forest Products, Inc. and the harvesting was carried out by Roseburg, the harvesting treatment accurately depicts typical harvesting conditions in small, non-fish-bearing streams in Oregon. The objectives of the Hinkle Creek stream temperature study were to identify and quantify changes to stream temperature patterns that occurred after the harvesting treatment was applied and to explain post-harvest stream temperature patterns with reach-level canopy closure data.

Changes to maximum, minimum and mean daily stream temperatures, diel temperature fluctuation and annual maximum seven-day mean temperatures were analyzed using repeated measures models that compared the mean pre-harvest relationship between temperatures observed in the harvested streams and temperatures observed in the unharvested streams to the post-harvest relationship. No significant changes to daily maximum stream temperatures were discerned when the overall response across the four harvested streams was considered, however after harvesting daily minimum and mean stream temperatures were significantly lower after harvesting, particularly on days when the minimum or mean temperature was above 12˚C. Diel stream temperature fluctuations increased significantly after harvesting, often to more than double the mean diel fluctuations that were observed before harvesting occurred. As there was no significant change to maximum daily temperatures, the increased diel range occurred because minimum daily temperatures decreased. There was no appreciable difference between annual maximum seven-day mean temperatures pre- and post-harvest. These results differ from *a priori* hypotheses that stream temperatures would become significantly warmer after harvesting.

Although change detection model results indicated no significant changes to maximum temperatures across the four streams, examination of individual reach responses illustrate that significant changes to maximum temperatures did occur in two of the streams, but because the streams responded divergently, no net changes were detected across the four streams. A closer examination of reach-level variables that could potentially affect stream temperature may partially explain the divergent and unexpected temperature responses. It is generally assumed that significant reductions in stream shading occur when the forest canopy is removed. However, a thick layer of organic logging slash partially covered the small streams one year after harvesting occurred and limited exposure of the streams to solar radiation. When cover due to logging slash was accounted for, only a mean 20% reduction in canopy closure occurred as a result of the harvesting treatment. This reduction is much lower than is generally assumed for streams that are clearcut without a vegetated riparian buffer. It is also likely that summer baseflows increased significantly following the harvest and that the greater volume of cooler water influenced stream heating. The combination of high levels of shade from the logging slash and high stream volumes during the post-harvest year may have prevented dramatic increases in maximum temperatures and caused minimum and mean temperatures to decrease.

The true impact of the harvesting treatment on summer stream temperatures in Hinkle Creek has likely yet to be observed. Over the next several years the protective layer of logging slash covering the harvested streams will decompose and as these watersheds are intensively managed with post-harvest herbicide treatments, it is probable that the streams will be exposed to high levels of solar radiation before the riparian canopy recovers. The balance between recovering riparian shade and volume of stream water will be crucial determinants of stream temperature patterns as these watersheds recover.

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Appendix A

Table A1. Regression line parameters for maximum daily stream temperatures in all stream pairs.

Stream pair	Year	Slope	Intercept
Fen	2002	0.98	12.8
Fen	2003	0.85	14.0
Fen	2004	0.92	14.3
Fen	2006	0.64	12.1
Clay	2002	1.42	13.7
Clay	2003	1.17	15.0
Clay	2004	1.25	14.5
Clay	2005	1.26	13.3
Clay	2006	1.27	15.2
Rus	2003	1.17	12.4
Rus	2004	1.05	12.0
Rus	2005	1.27	11.8
Rus	2006	1.17	12.7
BB	2002	0.80	13.0
BB	2003	0.77	12.6
BB	2004	0.87	12.9
BB	2005	0.82	13.0
BB	2006	1.11	13.6

Table A2. Regression line parameters for minimum daily stream temperatures in all stream pairs.

Stream pair	Year	Slope	Intercept
Fen	2002	0.91	11.7
Fen	2003	0.89	13.2
Fen	2004	0.92	13.4
Fen	2006	0.59	10.9
Clay	2002	1.31	12.2
Clay	2003	1.26	13.6
Clay	2004	1.27	13.6
Clay	2005	1.28	12.4
Clay	2006	1.08	12.4
Rus	2003	1.31	11.6
Rus	2004	1.14	11.2
Rus	2005	1.38	11.1
Rus	2006	0.98	10.9
BB	2002	1.43	12.2
BB	2003	1.33	12.4
BB	2004	1.21	12.0
BB	2005	1.40	11.8
BB	2006	1.05	12.1

Table A3. Regression line parameters for daily mean stream temperatures in all stream pairs.

Stream pair	Year	Slope	Intercept
Fen	2002	0.95	12.3
Fen	2003	0.89	13.6
Fen	2004	0.91	13.8
Fen	2006	0.62	11.5
Clay	2002	1.33	12.9
Clay	2003	1.24	14.2
Clay	2004	1.25	14.0
Clay	2005	1.27	12.8
Clay	2006	1.18	13.7
Rus	2003	1.27	12.0
Rus	2004	1.14	11.6
Rus	2005	1.36	11.5
Rus	2006	1.06	11.7
BB	2002	1.42	12.5
BB	2003	1.31	12.8
BB	2004	1.21	12.4
BB	2005	1.34	12.2
BB	2006	1.10	12.8

Figures A1-A6. The percent canopy closure before harvest (2004) and after harvest (2006) measured using a spherical densitometer and a digital camera (2006). The x-axis is the location of the sampling points along the stream's longitudinal profile. The zero position marks the downstream boundary of the harvest unit. The mean and standard deviations of percent canopy closure after harvest in harvested reaches are shown for data collected using a spherical densitometer and a digital camera.

Figure A1- Fenton Creek

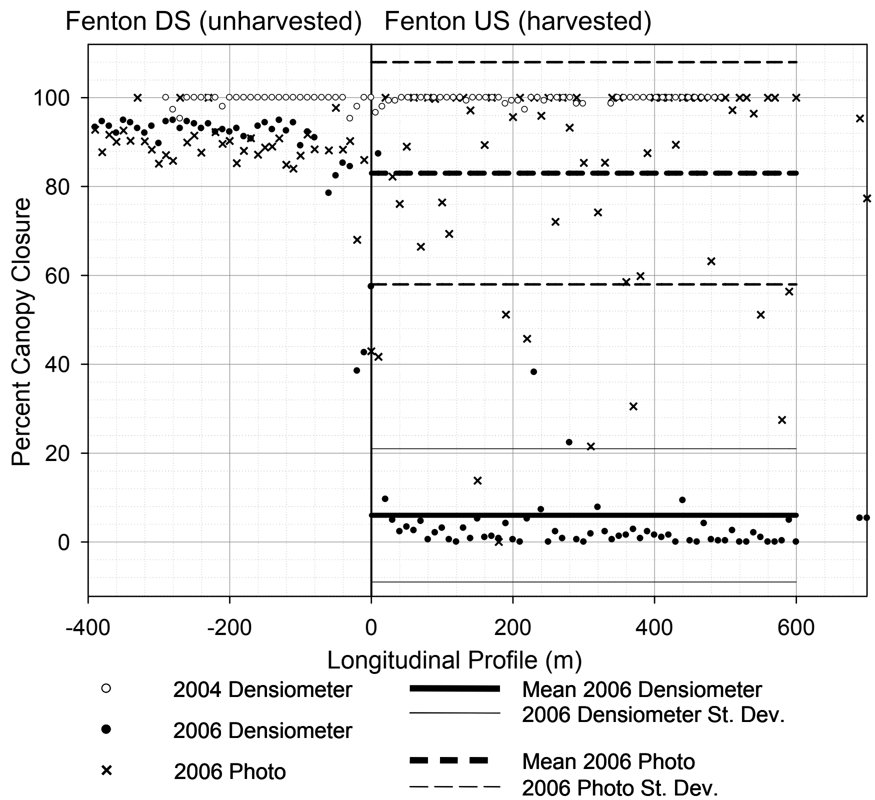

Figure A2- Clay Creek

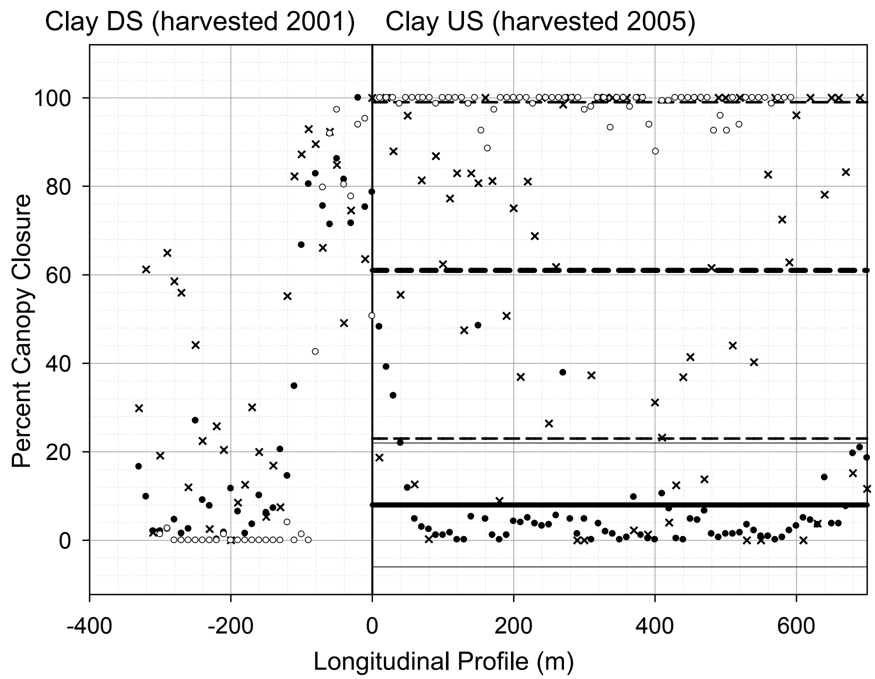

Figure A3- Russell Creek

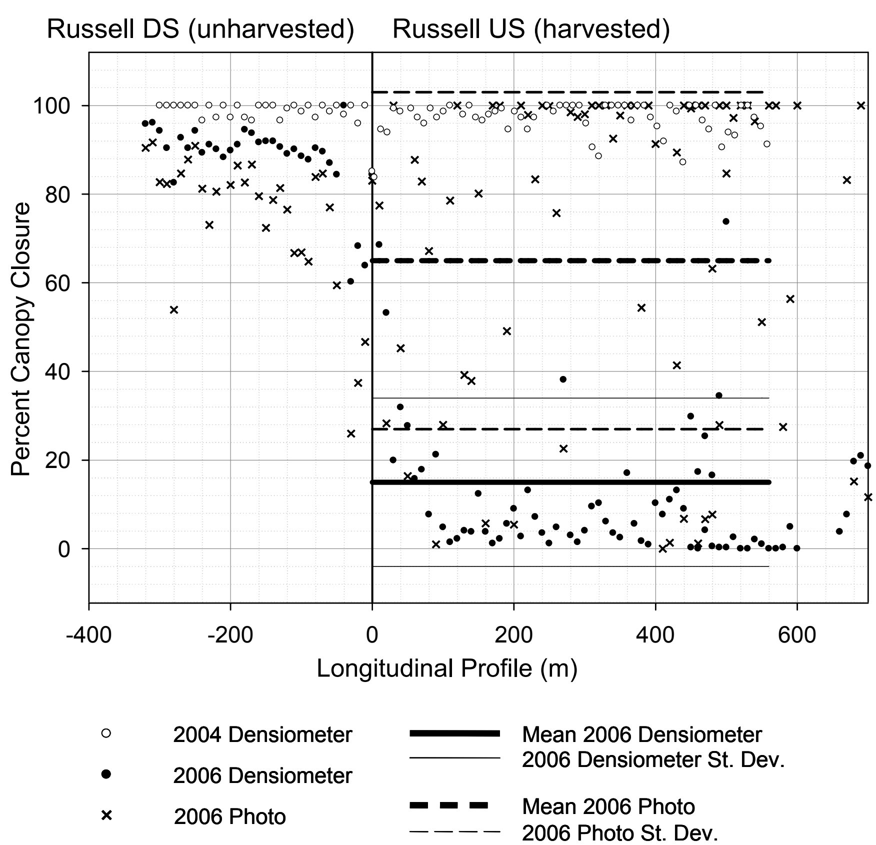

Figure A4- BB Creek

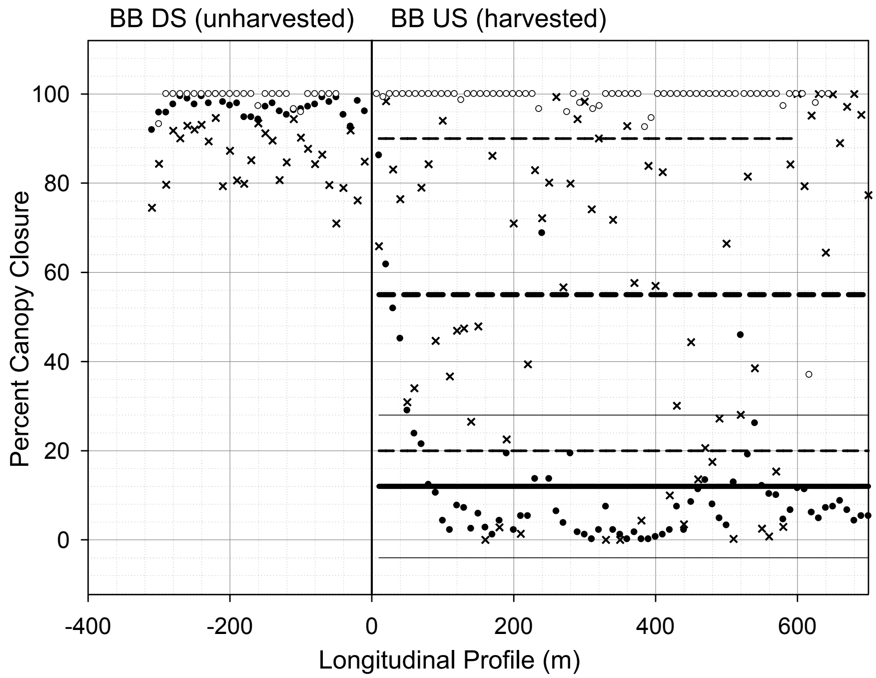

Figure A5- Myers Creek

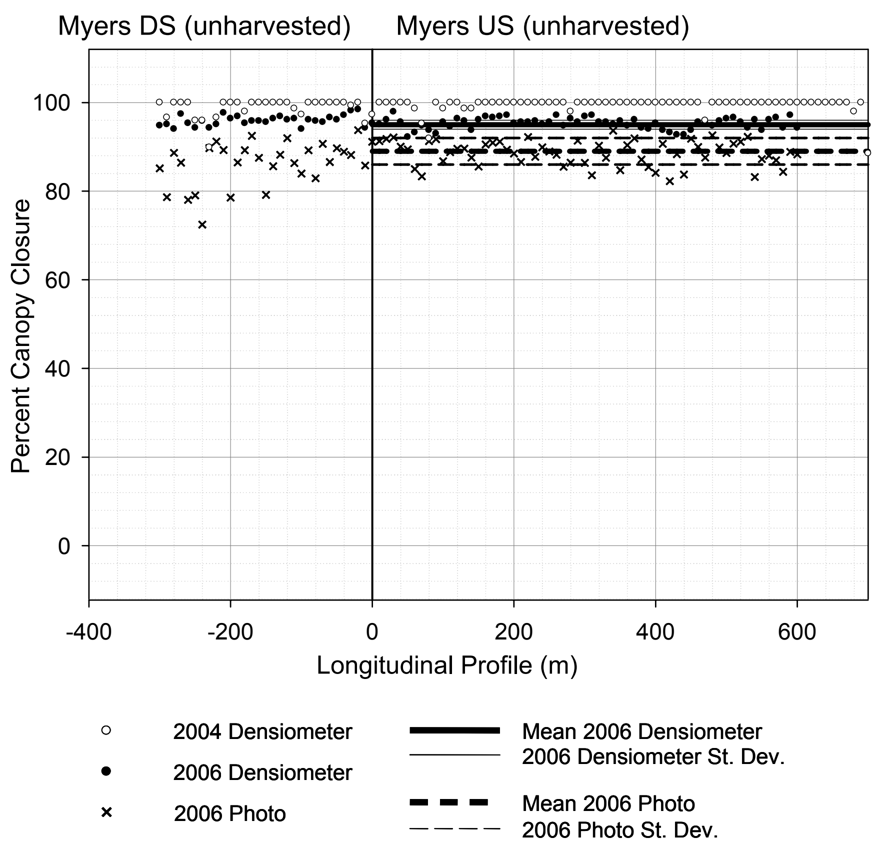

Figure A6- DeMerrseman Creek

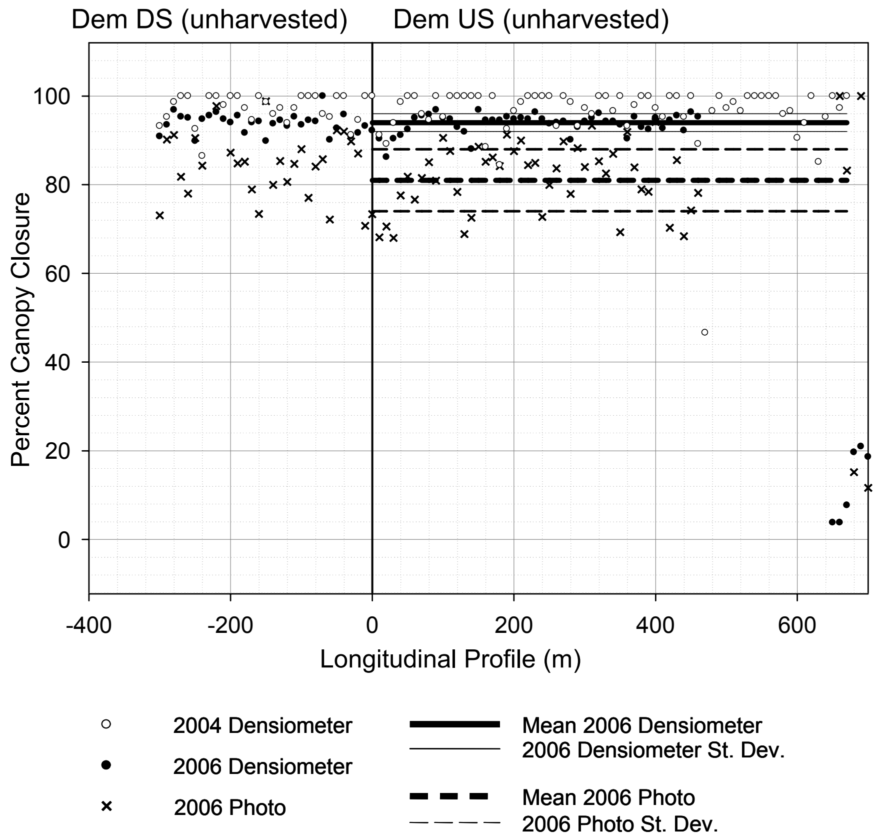

Figure A7. Daily minimum and maximum stream temperatures plotted in time series for Fenton Creek 2002-2006 and Myers Creek (unharvested) 2005.