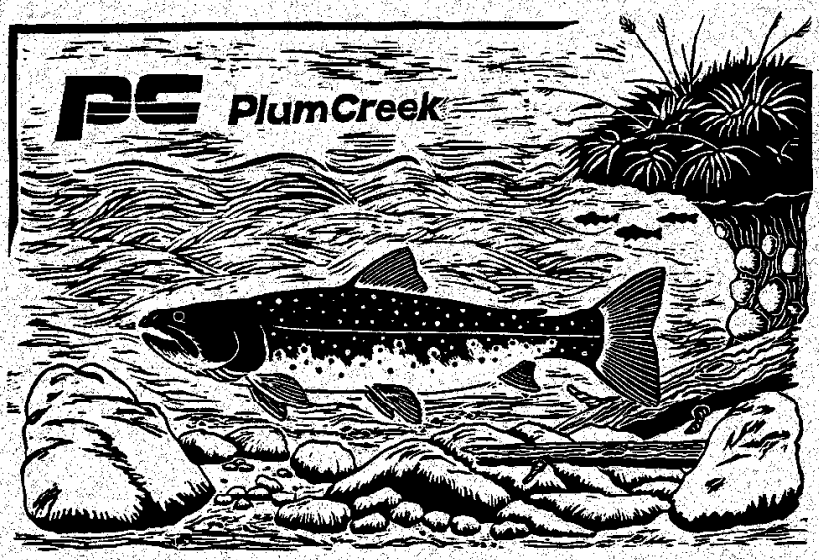
*Plum Creek Timber Company*
*Native Fish Habitat Conservation Plan*

# Stream Temperature Considerations in the Development of Plum Creek's Native Fish Habitat Conservation Plan

**Technical Report #12**

1998




PlumCreek

# STREAM TEMPERATURE CONSIDERATIONS IN THE DEVELOPMENT OF PLUM CREEK'S NATIVE FISH HABITAT CONSERVATION PLAN

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**Native Fish Habitat Conservation Plan**
**Technical Report #12**

Plum Creek Timber Company
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## ABSTRACT

This report addresses several questions related to the management of riparian areas for maintenance of stream temperatures suitable for native fish in Plum Creek's Habitat Conservation Planning Area. Subjects addressed included: 1) a review of temperature requirements of native fish in the planning area, including bull trout; 2) a review of winter conditions on trout in ice-covered streams; 3) a discussion of the potential influence of small streams on downstream fish-bearing waters; 4) results of a study of canopy cover before and after timber harvest along ten streams in Montana and Idaho; and 5) development of a predictive stream temperature model for western Montana and northern Idaho. The information described in this report will be used, in concert with other technical information, as a basis for developing a Habitat Conservation Plan for native salmonids on Plum Creek Timber Company's lands in the Columbia River Basin. Areas of significant uncertainty will be addressed through an adaptive management process being developed as part of the conservation plan.

## ACKNOWLEDGMENTS

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# 1.0 INTRODUCTION

It is well established that riparian timber harvesting can increase maximum stream temperatures (see review by Beschta et al. 1987). The primary mechanism for this stream warming is an increase in direct-beam solar radiation that reaches the stream surface when shading is removed (Brown 1969). Because bull trout are thought to be particularly sensitive to temperature changes (see reviews in Shepard et al. 1984; Goetz 1989; Fraley and Shepard 1989; Rieman and McIntyre 1993; Buchanan and Gregory 1997), Plum Creek Timber Company's Native Fish Habitat Conservation Plan must address potential temperature changes in streams.

To provide an intelligent foundation on which Habitat Conservation Plan (HCP) commitments can be developed, a number of questions must be answered. These include:

- What are the temperature requirements of bull trout and other native salmonids in the HCP area?
- What are the winter requirements of fish, and how can timber harvest effect their habitat during this time of year?
- Given that many streams in the planning area are non-fish bearing, to what extent can timber management along these streams influence downstream temperatures in fish-bearing waters?
- How do contemporary forest management activities (in compliance with existing regulations in Montana and Idaho) alter canopy cover (shade) over streams?
- Can a practical predictive model be developed to help facilitate management questions?

The purpose of this technical report is to address each of the questions posed above. Based on this technical report (and others), a technically-defensible HCP will be developed.

This document has seven sections. In ***Section 2***, existing information is reviewed on the temperature requirements of native fish in the HCP area, with emphasis given to bull trout requirements. New information that was collected from several streams draining Plum Creek Timber Company lands in Montana and Idaho is presented. This information, combined with that in the literature, offers a more complete picture of the temperature requirements of bull trout in Pacific Northwest streams. ***Section 3*** examines the effects of winter conditions on trout in ice-covered streams. This includes an examination of the behavior, habitat use, and mortalities of trout during winter. It also includes a brief description of the potential effects of timber harvest on trout habitat in winter. ***Section 4*** of

this report discusses the role of small, perennial, non-fish-bearing streams in contributing to temperatures in downstream fish-bearing waters, and what effect current regulations and forest practices may have on temperatures in these streams. ***Section 5*** presents an analysis of the effects of timber harvesting on canopy cover over perennial streams when that harvest has been conducted in compliance with Montana and Idaho forest practice regulations. Lastly, ***Section 6*** of this report describes several temperature prediction models that were developed based on temperature data at 100 sites in northern Idaho and western Montana. This model predicts maximum water temperatures as a function of stream reach elevation, canopy cover over the stream, and indices of yearly meteorologic conditions compared to long-term averages. Examples are presented of how nomographs developed from the prediction models might be used in a management, research, or monitoring context. ***Section 7*** summarizes the findings in this report.

# 2.0 TEMPERATURE REQUIREMENTS

In this section, we discuss water temperature requirements of salmonids in the Plum Creek Timber Company HCP project area. We focus attention on bull trout and show that temperatures required by bull trout adequately protect other native salmonids in the project area. We realize that factors other than temperature (e.g., physico-chemical stressors) can influence the spatial dynamics of salmonids both zoogeographically and locally, with some stresses acting at large ecoregion scales (e.g., effects of aridity) and others at much smaller scales (e.g., thermal gradients formed where groundwater mixes with surface water). Salmonid distribution in North America and worldwide, however, appears to be strongly linked to temperature (Power 1990).

Below we briefly cover several important aspects of temperature on the ecology of salmonids. First, we consider the physiological consequences of temperature on fish. This section is intended to give the reader background on the effects of temperature on fish energetics and production. Second, we offer a brief description of the various temperature metrics reported in the literature. We also identify relationships among the various metrics. Third, we briefly discuss the temperature requirements of bull trout by life stage. We then compare the temperature requirements of bull trout with other native salmonids that occur within the project area. Fourth, we discuss the effects of temperature on competitive interactions. Here, we focus on interactions between bull trout and other salmonids. Finally, we describe the existing water temperature standards.

## 2.1 PHYSIOLOGICAL CONSEQUENCES OF TEMPERATURE

Physiological effects of high or low temperature on fish are well known (Tytler and Calow 1985; Jobling 1994). Acclimation[1] and seasonal acclimatization[2] to temperature changes are important in the ecology of fish, although caution is needed in extrapolating laboratory studies to field habitats. For example, the absence of natural seasonal cues, such as changing photoperiod, may cause an artificially acclimated fish to respond differently than one that has been naturally acclimatized.

The effect of temperature on biochemical and physiological processes drives fish to select environmental temperatures at which they can function efficiently (Coutant 1987). Because different physiological processes (e.g., ingestion and metabolism) may have different optimal temperatures, the temperature selected by a fish often represents a compromise, or "integrated optimum." Fish

[1]Acclimation refers to the process of physiological adjustment, typically to a single parameter such as temperature, under artificial conditions.

[2]Acclimatization refers to adjustments made under natural environmental conditions, including seasonal changes in temperature, photoperiod, and associated hormones.

appear to select temperatures that maximize the amount of energy available for activity and growth, or metabolic scope (the difference between standard and maximum metabolic rates) (Fry 1971; Jobling 1994). Certainly, habitat selection in the wild involves a compromise between temperature requirements and other important factors, such as food availability and avoidance of predators and competitors (Coutant 1987).

Brungs and Jones (1977) indicate that the most sensitive physiological function appears to be growth rate, which is an integrator of all physiological responses. The rate of growth at various temperatures is a function of ingestion and metabolism (Jobling 1994). Under conditions of unlimited food, an increase in temperature will result in an increase in food intake, but at high temperature there will be an abrupt decline in ingestion rates. Metabolic rate, on the other hand, increases with increasing temperature. The temperature at which the difference between ingestion rate and metabolic rate is maximum is called the *optimum temperature for growth.* For most salmonids, the optimum temperature for growth occurs between 10° and 17°C (Table 2.1).

Extreme temperatures (both low and high) can lead to death. Proteins, including the enzymes that catalyze critical biochemical reactions, are temperature sensitive. High temperatures can cause structural degradation (denaturation), resulting in partial or complete loss of function. Death can occur quickly or may be delayed. The temperature at which a fish succumbs to thermal stress depends on the temperature to which it was acclimatized and on its developmental stage (e.g, embryo, fry, juvenile, adult). Fish that experience changing environmental temperatures, however, have cellular and subcellular mechanisms for adapting to the new conditions. Many physiological adjustments result from switching on or off genes that are responsible for the manufacture of particular proteins (Jobling 1994). For example, some salmonids (e.g., brown trout, cutthroat trout, and chinook salmon) under heat stress initiate the synthesis of *heat shock proteins* (HSPs) (Fader et al. 1994). These reconfigure proteins that become denatured at higher temperatures, thereby allowing them to function biochemically. In addition, fish may produce alternate enzymes or *isozymes* to catalyze the same reaction more efficiently at different temperatures (Jobling 1994).

## 2.2 TEMPERATURE METRICS

Because there are several temperature metrics (units) that are used in the literature, we side-step the general flow of the report to define these metrics. In this section, we define the most commonly used metrics, describe how temperatures are often recorded, and examine relationships among the various temperature metrics.

### 2.2.1 Definitions

Here we offer definitions of the most commonly used metrics. Metrics that describe average temperature conditions may be based on two or more instantaneous measurements. Clearly, the

greater the number of instantaneous measurements, the more valid the mean or average measurement.

**Instantaneous or Snap-Shot Temperatures**--These are water temperatures recorded at a specific point in time.

**Daily Average Temperature**--This is the mean water temperature for a given 24-hour period.

**Daily Maximum Temperature**--This is the highest water temperature recorded during a given 24-hour period.

**Daily Minimum Temperature**--This is the lowest water temperature recorded during a given 24-hour period.

**Maximum Daily Average Temperature**--This is the warmest daily average water temperature recorded during a given year or survey period.

**Maximum Daily Maximum Temperature**--This is the warmest daily maximum water temperature recorded during a given year or survey period.

**Maximum Weekly Average Temperature (MWAT)**--This is the mean of daily average water temperatures measured over the warmest consecutive seven-day period (typically during a given year).

**Maximum Weekly Maximum Temperature (MWMT)**--This is the mean of daily maximum water temperatures measured over the warmest consecutive seven-day period (typically during a given year).

**Monthly Mean Temperature**--This is the average water temperature recorded during a given month.

### 2.2.2 Temperature Recording Devices

Water temperatures in bull trout streams have been measured with a number of different recording devices. For example, Bonneau and Scarnecchia (1996), Pratt (1984), and Heimer (1965) used hand-held thermometers to record water temperatures in bull trout streams. Saffel and Scarnecchia (1995), Adams (1994), Fraley et al. (1981), and Graham et al. (1980) used maximum-minimum thermometers to measure water temperatures, while Swanberg (1996), Thurow and Schill (1996), Parkinson and Haas (1996), Saffel and Scarnecchia (1995), Adams (1994), Riehle (1993), Martin

et al. (1992), Ziller (1992), Weaver and White (1985), Fraley et al. (1981), and Graham et al. (1980) used electronic thermographs to monitor temperatures in bull trout streams. Electronic thermographs consisted of Omnidata recorders (Adams 1994; Riehle 1993), Ryan Tempmentors (Adams 1994; Riehle 1993; Saffel and Scarnecchia 1995; Ziller 1992), Onset loggers (Adams 1994; Parkinson and Haas 1996; Swanberg 1996), and Foxboro loggers (Weaver and White 1985).

Hand-held temperature recorders are often used to measure water temperatures during fish or stream habitat surveys. For example, it appears that Pratt (1984) and Thurow (1987) recorded instantaneous water temperatures with hand-held recorders during their fish surveys. Because the accuracy of snorkel counts can be influenced by water temperature (Hillman et al. 1992), observers frequently record instantaneous water temperatures with hand-held recorders before conducting snorkel surveys. As indicated by the name, maximum-minimum thermometers record the highest and lowest water temperatures during a given period. The length of the sample period varies with the frequency that an observer reads the thermometers. For example, Saffel and Scarnecchia (1995) read their maximum-minimum thermometers every two weeks, while it appears that Adams (1994) read hers daily.

Electronic thermographs can be programmed to record temperatures at various intervals throughout a survey period. Adams (1994) programmed her thermographs to record temperatures every 48 or 60 minutes. Thermographs used by Swanberg (1996), Thurow and Schill (1996), and Parkinson and Haas (1996) recorded temperatures hourly. Most researchers, however, do not report the frequency that thermographs record water temperatures. Additionally, they rarely report the error associated with electronic thermographs. This error may be quite large. For example, Ozaki (1998) found that during calibration tests, thermographs at any one time differed in temperature by about two degrees. She also noted that there was a significant difference in mean temperatures between data loggers and also in mean temperatures within individual types of loggers.

### 2.2.3 Relationships Among Temperature Metrics

We examined the relationships among various temperature metrics by compiling 225 temperature records from 73 streams in Montana and Idaho. Data were collected by Plum Creek Timber Company, Potlatch Corporation, and the Idaho Division of Environmental Quality. Approximately 70 of the 225 records were at sites that supported bull trout. Onset temperature loggers recorded water temperatures at 30-minute intervals (records the average temperature within the 30-minute interval) during the summers of 1994 to 1997. For each site, we calculated the maximum daily maximum temperature, maximum daily average temperature, MWMT, MWAT, mean July temperature, and mean August temperature. For each metric, we calculated the mean, standard deviation, maximum, and minimum values. We then used scatter plots and a Pearson correlation matrix to assess relationships among temperature metrics. Finally, we used simple linear regression to describe the relationships among the temperature metrics.

Daily maximum temperatures were on average 1° to 5°C warmer and more variable than other temperature metrics (Table 2.2). Mean July and August temperatures were consistently colder and less variable than mean values of the other metrics. The average difference between maximum daily maximum and maximum daily average temperature was about 3°C. Similarly, the mean difference between MWMT and MWAT was about 3°C. On average, there was only about a 1°C difference between maximum daily maximum temperature and MWMT, while there was less than a 1°C difference between maximum daily average temperature and MWAT.

Not surprising, we found significant relationships among all combinations of temperature metrics. Pearson correlation coefficients consistently exceeded 0.89 for all possible combinations of temperature metrics (Table 2.3). In addition, all relationships were linear (Figures 2.1-2.5). Simple linear regression models explained 84 to 99% of the variation between independent and dependent temperature metrics (Figures 2.1-2.5). For example, the linear model, MWMT = 1.15(MWAT) + 0.41, explained 90% of the variation between MWMT and MWAT (Figure 2.5). These models indicate that a 10°C MWMT corresponds to a 10.6°C daily maximum, 9.3°C maximum daily average, 8.0°C mean July temperature, 7.9°C mean August temperature, and 8.7°C MWAT.

These relationships comport with those in the literature. For example, the ODEQ (1995) notes that in Oregon streams, maximum water temperatures are typically recorded in July or August when incoming solar radiation levels are high, air temperatures are high, days are long, and stream flows are low. The ODEQ also notes that MWMT will nearly always be slightly lower than the single warmest daily maximum temperature. They report, as an example, that the MWMT is 1.5°C lower than the daily maximum temperature for the warmest single day on the Grande Ronde River. This is close to the 0.8°C mean difference we found between these two metrics in 73 streams in Montana and Idaho (Table 2.2).

## 2.3 SALMONID TEMPERATURE REQUIREMENTS

In this section, we describe the temperature requirements of native salmonids in the HCP project area. Because bull trout are the major thrust of this document, we focus mostly on bull trout temperature requirements. We evaluate in some detail the temperature requirements of various life stages of bull trout. We also offer additional or new information on relationships between bull trout and temperature. We conclude this section by comparing bull trout temperature requirements with those of other native salmonids in the project area.

Many biologists believe that bull trout are quite sensitive to temperature change (see reviews in Shepard et al. 1984; Goetz 1989; Fraley and Shepard 1989; Rieman and McIntyre 1993; Buchanan and Gregory 1997). The fact that bull trout are frequently found in cold water streams has lead many biologists to conclude that bull trout are "cold stenotherms." This means that optimal survival and reproduction of bull trout occurs within a narrow range of cold temperatures. According to Reiman

and McIntyre (1993), the evidence is mostly correlative, leaving critical thermal thresholds poorly defined. Indeed, regulatory agencies have established temperature criteria for bull trout streams (see Section 2.5) with little information or knowledge of the influence of temperature on bull trout distribution. This is not intended as a criticism of the agencies, but rather as an acknowledgment of our lack of understanding of bull trout habitat requirements.

There is virtually no information on the physiological optimum temperature or temperature tolerance of bull trout. Plum Creek Timber Company has allocated funds for studies that will assess the growth and survival temperature criteria for bull trout and identify the presence and function of heat shock proteins in bull trout. Results from these studies will offer useful information on bull trout temperature requirements. In the absence of these data, we must rely on the correlative data available.

### 2.3.1 Adult Bull Trout Rearing and Migration

It is quite difficult to separate temperature requirements for rearing and migrating adult bull trout. This is in part because the researcher does not always know if the adult is rearing or staging during migration. In addition, adult rearing temperatures may vary among life history types. For example, resident, head-water adults may occupy colder sites than migratory adults. Nevertheless, some researchers have attempted to describe separate temperature requirements for adult rearing and migration.

The literature indicates that adult bull trout can rear within a wide range of temperatures. Buchanan and Gregory (1997) note that adult bull trout are present at temperatures that range from 4° to 18°C. Other studies have reported bull trout at water temperatures approaching 20°C (e.g., Thurow 1987; Saffel and Scarnecchia 1995; Swanberg 1996). Adams (1994) observed adult bull trout in sites with daily maximum water temperatures of 20.5°C in the Weiser River drainage. These observations occurred several times during a two-year study. Bull trout densities were also relatively high in these sites (3.9-5.1 trout/100 $m^2$). Although Buchanan and Gregory (1997) suggest that adults prefer temperatures of 9° to 13°C, we can find no evidence for a preferred temperature range. Our review of the literature indicates that adult bull trout can rear at water temperatures ranging from 4 to 20.5°C.

Because adult migration can occur at nearly anytime during the year, it is difficult to identify temperatures required for migration. Buchanan and Gregory (1997) believe that temperatures required for adult bull trout migration range from 10° to 12°C. Work by Swanberg (1996) in the Blackfoot River basin indicates that bull trout migrate at temperatures that range from 12° to 20.0°C (mean=17.7°C). Swanberg (1996) notes that most (73%) bull trout began migrating during temperature peaks of 18° to 20°C. In Rapid River, a tributary of the Salmon River, bull trout migrated at temperatures of 10° to 15°C (Elle and Thurow 1994; Elle 1995). These studies indicate

that adult bull trout migrate at daily temperatures that range from 10° to 20°C. It is likely that adult bull trout also migrate at temperatures lower than 10°C.

### 2.3.2 Bull Trout Spawning

Given suitable gravels and cover, bull trout appear to select groundwater upwelling zones for spawning (Heimer 1965; McPhail and Murray 1979; Fraley and Shepard 1989; Riehle 1993; WWP 1995). These spawning areas often occur in low-gradient sections of third or fourth order streams. Spawning usually occurs just downstream from low and high gradient interfaces, or aggrading areas (Graham et al. 1980). Use of these areas may have survival advantages if the water quality (suitable temperatures and dissolved gases, and lack of heavy metals and sediments) in such areas is more suitable than in areas without groundwater. For fall-spawning fish like bull trout, upwelling zones may protect developing embryos and fry from winter ice scouring and freezing (see Section 3.0).

Because bull trout spawn during fall (descending limb of the thermograph) and often select nest sites in groundwater upwelling zones, water temperatures at time of spawning are generally quite cool. The ODEQ (1995), Buchanan and Gregory (1997), and EPA (40 CFR 131.E.1-(1997)) believe that the optimal temperatures for bull trout spawning range from 4° to 10°C. Several studies support this temperature range (e.g., Riehle 1993; McPhail and Murray 1979). Other studies have shown that bull trout can spawn at water temperatures slightly higher than 9° or 10°C. For example, Heimer (1965) found bull trout spawning in or near upwelling zones with water temperatures of 9° to 12°C in an artificial channel in the Clark Fork River. WWP (1995) also found bull trout spawning in the Clark Fork River at water temperatures at most redd sites ranging from 10° to 11°C. The EPA reports that some spawning occurred in the Flathead River system in water temperatures as high as 12°C. These studies indicate that bull trout spawning can occur in water temperatures ranging from 4° to 12°C.

### 2.3.3 Bull Trout Egg Incubation

Bull trout require about 350 to 440 temperature units[3] (°C) after fertilization to hatch (Weaver and White 1984; Gould 1987). EPA (40 CFR 131.E.1 (1997)) notes that the optimum incubation temperature appears to be 4°C; ODEQ (1995) and Buchanan and Gregory (1997) believe that optimal egg incubation temperatures for bull trout range from 1° to 6°C. Work by McPhail and Murray (1979) and Weaver and White (1985) supports a 1° to 6°C temperature range for bull trout incubation. The studies by McPhail and Murray (1979) also indicate that warmer temperatures tend to increase the rate of embryonic development and yolk absorption and decrease alevin size and survival. Brannon (1987) notes that some salmonids compensate or change their development rate

[3]One temperature unit equals one degree C above 0°C for one 24-hr period.

per unit of temperature as temperatures change. That is, more thermal units are required for hatching at high temperatures than at low temperatures. Without such compensation, if water temperatures were unusually warm during development over the winter, fry emergence in the spring would occur earlier than normal, and fry survival would probably be reduced. This aspect of bull trout ecology deserves more study.

### 2.3.4 Juvenile Bull Trout Rearing

As with adults, juvenile bull trout rear within a wide range of temperatures. Researchers have found juveniles rearing at temperatures as low as 4°C (*sensu* Buchanan and Gregory 1997) and as high as 20.5°C (Adams 1994). Although the findings of Adams (1994) seem relatively high, others have also found juvenile bull trout at temperatures near 20°C. For example, Saffel and Scarnecchia (1995) observed juvenile bull trout in tributaries of Lake Pend Oreille at temperatures of 20°C. In tributaries of the South Fork Salmon River, Thurow (1987) reported temperatures up to 19.5°C in some sites with juvenile bull trout. As noted by Rieman and McIntyre (1993), most studies have found juvenile bull trout at temperatures less than 15°C. This has lead many to believe that temperatures in excess of 15°C may limit bull trout distribution.

In an attempt to better understand relationships between juvenile bull trout distribution and water temperatures, we compiled stream temperatures from 68 to 70 sites in 33 streams supporting bull trout on Plum Creek Timber Company lands in Montana and Idaho. Onset temperature loggers recorded water temperatures at 30-minute intervals (records the average temperature within the 30-minute interval) during the summers of 1994 to 1997. For each site, we calculated the maximum daily maximum temperature, maximum daily average temperature, MWMT, MWAT, mean July temperature, and mean August temperature. For each metric, we calculated the mean, standard deviation, maximum, and minimum values (Table 2.4). These data indicate that for all metrics, mean temperatures among 33 bull trout streams were less than 15°C. However, maximum values for each metric, save monthly values, exceeded 15°C. The mean MWMT for the 33 bull trout streams was 12.4°C. The maximum MWMT was 18.7°C.

These data and those reported in the literature demonstrate that juvenile bull trout exist in streams with temperatures ranging between 4 and 20.5°C. Although there are no laboratory data, field studies (Plum Creek Timber Company unpublished data; Mullan et al. 1992; Saffel and Scarnecchia 1995) seem to indicate that optimal juvenile bull trout rearing temperatures fall between 10 and 15°C.

## 2.3.5 Comparison of Salmonid Temperature Requirements

In the sections above we described the temperature ranges that bull trout have been found to occupy. Where data are available, we also identified optimal temperature ranges. In this section, we compare

bull trout temperature requirements with those of other native salmonids in the HCP project area. Our intent here is not to give an exhaustive review of temperature requirements for each life stage of each salmonid species in the project area. Rather, our purpose is to summarize temperature requirements of other salmonids in the project area and compare those to bull trout.

Other native salmonids observed within the Plum Creek Timber Company project area include westslope cutthroat trout, redband trout, steelhead, chinook salmon, and mountain whitefish. Numerous studies have evaluated the temperature requirements of these species. Some work includes both laboratory and field studies. We relied mostly on the work of Barnhart (1991, in Stoltz and Schnell 1991), Bell (1991), Coutant (1977), Lanz (1971, in Beschta et al. 1987), Mantelman (1960, in Beschta et al. 1987), Daily (1971), Meehan and Bjornn (1991), Northcote and Ennis (1994), Orcutt et al. (1968), Wydoski and Whitney (1979), McIntyre and Reiman (1995), Pennell and Barton (1996), and Bjornn and Reiser (1991).

Before we continue, it is necessary to define some frequently used terms. An *optimum* temperature is generally the temperature where growth and metabolic efficiency are maximized (see discussion in Section 2.1). Because physiological optima are affected by and interact with acclimation temperatures, temperature optimums often are a "zone of efficient operation," rather than a single temperature value (Crawshaw 1977). *Selected* or *preferred* temperatures are temperature ranges in which fish congregate or spend the most time, assuming choice is available (Reynolds 1977). Preferred temperatures are generally within the range of optimum temperatures. *Avoidance* temperatures are those where a fish will actively choose other temperature conditions, if choice is available (Reynolds 1977).

We find that adult bull trout can rear and migrate within a wide range of temperatures (4-20.5°C). This range is similar to those observed during steelhead migration (<21°C; Table 2.5). Temperatures at the time of bull trout spawning range from 4 to 12°C, which is also similar to other fall-spawning salmonids in the project area (Table 2.5). Because mountain whitefish are late-fall spawners, they tend to spawn at colder temperatures than both chinook salmon and bull trout. Steelhead, redband trout, and westslope cutthroat trout spawn during spring at temperatures between 2 and 20°C. As with chinook and bull trout, optimal spawning temperatures for steelhead, redband trout, and westslope cutthroat trout tends to be ≤10°C. Incubation temperatures for other native salmonids in the project area are warmer than those for bull trout. Incubation temperatures for chinook salmon, a fall spawner, range from 5 to 14.4°C. In comparison, incubation temperatures for bull trout range from 1 to 6°C.

Juvenile bull trout can rear within a wide range of water temperatures (4-20.5°C); however, optimal temperatures appear to fall between 10 and 15°C. This optimal temperature range appears to be cooler than that for other native salmonids in the project area (Table 2.5). For example, steelhead, rainbow trout, and chinook all have optimal temperatures that range from 10°C to well above 15°C.

The upper optimal temperature for rainbow trout is 22°C. Juvenile steelhead avoid temperatures greater than 22°C. The optimal temperature range for cutthroat trout (12-15°C) falls within the optimal temperature range for bull trout. In addition, preferred temperatures for native salmonids in the project area are similar to the optimal temperatures of bull trout (Table 2.5).

In summary, water temperatures occupied by bull trout are within or cooler than those used by other native salmonids in the HCP project area. Managing water temperatures suitable for bull trout should not negatively affect other native salmonids if the latter are not sensitive to or stressed at cooler temperatures.

## 2.4 TEMPERATURE EFFECTS ON COMPETITIVE INTERACTIONS

In this section we focus on temperature effects on interactions between bull trout and introduced (non-native) brook trout. This is because most work shows that bull trout are less likely to coexist compatibly with brook trout than with other salmonids. Indeed, there is little evidence that bull trout compete with other salmonids within the project area. For example, Pratt (1984) studied the habitat use and interactions of juvenile cutthroat trout and bull trout in the upper Flathead River basin and found that they used specifically different habitat when together or separate (selective segregation).

Competitive interactions among fluvial salmonids usually translate into attempts by individuals of the same or different species to secure territories for adequate space and, therefore, food or cover or both (Chapman 1966). Some studies have shown that competition among various fish species can be mediated or controlled by water temperature (Baltz et al. 1982; Reeves et al. 1987; Hillman 1991; De Staso and Rahel 1994). For example, De Staso and Rahel (1994) examined the influence of water temperature on interactions between juvenile cutthroat trout and brook trout in a laboratory stream. They note that the two species were nearly equal competitors at 10°C, but brook trout showed a clear competitive dominance over cutthroat trout at 20°C.

The ODEQ (1995) generalized this knowledge to interactions between bull trout and brook trout. The ODEQ state, "*A factor greatly complicating bull trout temperature requirements, however, is competition with brook trout, an introduced species. This competition occurs today in approximately one-fourth to one-third of the bull trout habitat. Brook trout out-compete bull trout at all but the lowest temperatures.*" As we demonstrate below, there are a number of correlative studies that suggest interactions occur between brook and bull trout; however, we found no studies that address the influence of water temperatures on these interactions.

Several observations suggest that competition occurs between bull and brook trout. For example, Wallis (1948) noted that the feeding habits of brook trout and bull trout were similar. Both species consumed primarily aquatic insects, although brook trout took a slightly higher percentage of

terrestrial insects. Rode (1988) believed that stocking brook trout into McCloud Reservoir in California may have contributed to the demise of bull trout there. In all areas in Montana where bull trout and brook trout occur together, bull trout populations have declined (Goetz 1989). Dambacher et al. (1992) assessed the distribution, abundance, and habitat use of bull trout and brook trout in Sun Creek, Crater Lake National Park. They found that both species used similar habitat types and microhabitats. Both preferred pools over other habitats, but brook trout appeared to dominate pool inlets. Dambacher et al. (1992) concluded that competition and hybridization threaten the bull trout with extinction in Sun Creek. Parkinson and Haas (1996) suggested that bull trout and brook trout were segregated by temperature in the Mesilinka and Osilinka rivers. Given that these studies are correlative, they do not demonstrate cause-and-effect relationships between bull trout and brook trout.

Recently, Nakano et al. (in press) examined competitive interactions for foraging space among brook trout, native bull trout, and westslope cutthroat trout in Elk Creek, a tributary of the Swan River, Montana. In this study, which was more experimental than correlative, the authors found that when the three species were together, they interacted with each other, forming a size-structured, mixed-species dominance hierarchy. When cutthroat were removed, brook trout increased foraging rates and distances and used cover less. Bull trout did not change behavior. When the authors removed brook trout, bull trout increased foraging rates and distances and occupied more exposed positions. Based on these observations, Nakano et al. (in press) suggested that competitive interactions with brook trout are an important factor in regulating bull trout densities. This study, although more experimental than others, does not establish that interactions are influenced by water temperatures.

Not all studies suggest that bull trout and brook trout interact for food and space. For example, work by Rich (1996) in the Bitterroot basin found little evidence that brook trout were replacing bull trout. He noted that different habitat requirements between the two species appeared to be the most important factor influencing their distributions. Brook trout in the Bitterroot system occupied streams having habitat conditions where bull trout were normally absent. Clancy (1993) also reported differences in the habitats used by brook and bull trout. Perhaps these studies demonstrate what Connell (1980) referred to as the "*ghost of competition past*." That is, the two species are segregated today because of past interactions.

With few exceptions, the literature seems to suggest that introduced brook trout interact with native bull trout, although these studies are mostly correlative (i.e., they do not demonstrate cause-and-effect relationships). However, we cannot find where the literature demonstrates that temperature influences the outcome of interactions between the two species. Given that brook trout have a wide optimum temperature range (7.0-20.3 °C; Table 2.1) and they seek groundwater upwelling sites for spawning, it may be that brook trout can displace bull trout even at cold temperatures. The work of Cavallo (1997) demonstrates this point. He found that cold water temperatures in springbrooks in the Middle Fork Flathead River did not prevent brook trout from invading and displacing native

salmonids such as cutthroat and bull trout. Cavallo (1997) concludes that cold temperatures alone will not prevent brook trout from invading streams supporting cutthroat and bull trout.

## 2.5 EXISTING WATER TEMPERATURE STANDARDS

Agencies establish water quality standards for the purpose of protecting the beneficial or designated uses of waters. Under the federal Clean Water Act, states are to specify the appropriate uses of water and adopt water quality criteria that protect the designated uses. A water quality standard, as defined by the Clean Water Act, has two elements. The first element is the beneficial use being protected. The second is the water quality criterion, which represents the quality of water that supports a particular use. Criteria are expressed as numeric or narrative statements. Narrative criteria define limits, often by stating that waters shall be maintained free from some condition. Numeric criteria require the concentration or value of certain chemical or physical water characteristics, such as temperature, to be maintained above or below specific levels. Standards may be the same for all waters in the state, or they may vary from basin to basin.

Temperature standards are currently written as an amount of increase in water temperature allowed because of anthropogenic activity. The amount of increase allowed may vary by basin and the background temperature of the water. When water temperatures are at or above a specified value, no measurable increase in temperature due to human activity is allowed. Temperature standards specific to bull trout have been written for streams in Idaho and Oregon. Montana and Washington do not have standards written specifically for bull trout streams, although Washington recently proposed standards for protecting aquatic life (Hicks 1998). Temperature standards for bull trout streams were written based on the assumption that bull trout are cold stenotherms. Below we describe existing and proposed standards for bull trout streams. It is unknown whether or not these standards would protect other native salmonid species in the HCP project area.

### 2.5.1 Oregon Department of Environmental Quality (ODEQ)

The ODEQ has established a temperature standard for bull trout streams in Oregon (ODEQ 1995). That standard states that, "*Waterbodies serving as habitat to Bull Trout should not exceed maximum temperatures higher than 50°F* [10°C]." This criterion is measured as a rolling 7-day average of daily maximum temperatures (MWMT). ODEQ states further that, "*If the 7-day average of daily maximum temperatures exceeds 50°F* (10°C), *the stream and riparian conditions would be required to be restored or allowed to return to the most unaltered condition feasible for the purpose of attaining the coldest stream temperatures possible under natural background conditions.*" The ODEQ note that MWMT will nearly always be slightly lower than the single warmest daily maximum temperature during the seven-day period.

### 2.5.2 U.S. Environmental Protection Agency (EPA)

The EPA issued a site-specific temperature criterion for those waterbody segments in Idaho where bull trout spawn and juvenile bull trout rear (40 CFR 131.E.1.i.d (1997)). "*This Rule establishes a maximum weekly maximum temperature (MWMT) criterion of 10°C for the months of June, July, August and September for the protection of bull trout spawning and juvenile rearing in natal streams, expressed as an average of daily maximum temperatures over a consecutive 7-day period.*" EPA believes that these criteria will adequately protect the most critical and vulnerable life states of bull trout.

The EPA selected MWMT over a consecutive 7-day average of daily averages (MWAT) for several reasons. First, greater diurnal fluctuations around the mean daily temperature can be one effect of intensive watershed management (e.g., loss of riparian vegetation). Second, the available literature is insufficient to derive temperature criteria to be protective of short-term temperature extremes (e.g., daily maxima). Finally, MWMT is consistent with other temperature criteria that have been established or recommended to protect bull trout (ODEQ 1995; USFS INFISH).

### 2.5.3 Idaho Division of Environmental Quality (IDEQ)

The IDEQ responded to the EPA temperature standard with an alternative temperature standard for bull trout habitat in Idaho. The IDEQ proposed that, "*Water temperatures for the waters identified under Section 250.02.e.i. shall not exceed twelve degree Celsius (12°C) daily average during June, July and August for juvenile bull trout rearing, and nine degrees Celsius (9°C) daily average during September and October for bull trout spawning. For the purposes of measuring these criteria, the daily average shall be generated from a recording device with a minimum of six (6) evenly spaced measurements in a twenty-four (24) hour period.*" The IDEQ further proposed, "*Exceeding the bull trout temperature criteria will not be considered a water quality standards violation when the air temperature exceeds the ninetieth (90th) percentile of the seven (7) day average daily maximum air temperatures for the warmest seven (7) day period of the year.*" The IDEQ believe this standard is easier to obtain than a weekly average, and that the daily average takes into account the overall thermal regime of the water body and more accurately reflects the overall daily heat load to aquatic organisms.

### 2.5.4 Washington Department of Ecology (WDE)

The WDE has recently issued a preliminary review draft discussion paper that evaluates use-based temperature standards for protecting aquatic life (Hicks 1998). They propose that during the beginning of spawning (1-15 September), bull trout and Dolly Varden waters cannot exceed a MWMT of 8-9°C or a single daily maximum temperature of 10°C. During peak spawning, these waters cannot exceed a MWMT or single daily maximum temperature of 6-7°C and 8°C,

respectively. Water temperatures during juvenile rearing cannot exceed a MWMT or single daily maximum temperature of 10-12°C and 13°C, respectively. Finally, during adult migration, streams cannot exceed a MWMT or single daily maximum temperature of 15-16°C and 22°C, respectively. As we indicated, these preliminary use-based temperature standards are in review.

# 3.0 WINTER REQUIREMENTS

In section 2.0 we discussed temperature requirements of bull trout and other salmonids in the HCP project area. That discussion focused primarily on temperature requirements during warmer, ice free stream conditions (i.e., spring, summer, and fall). In this section we examine the effects of winter conditions on trout in ice-covered streams. Specifically, we examine behavior, habitat use, and mortalities of trout during winter. Because bull trout and many other salmonids typically live in streams with diverse icing conditions, we also describe those conditions and assess how ice affects trout habitat. Lastly, we briefly describe the effects of timber harvest on trout habitat in winter.

Before we describe the winter ecology of trout, we need to define "winter." Winter is the period from egg deposition by autumn-spawning salmonids (and coincident with a decline in water temperature) to the loss of all surface ice (often accompanied by snowmelt), and before any reproductive activity by spring-spawning fish. According to Cunjak (1996), this definition has more biological relevance than one that follows calendar dates. Because most streams supporting juvenile bull trout are subject to near-freezing temperatures (<1 °C) and ice-formation during the winter, we reviewed studies conducted in streams subject to those winter conditions.

## 3.1 WINTER BEHAVIOR

When water temperatures decline below about 10°C, salmonids change behavior from mostly feeding and defending territories to hiding and schooling (Bjornn and Reiser 1991; Hillman et al. 1992). This involves movement to areas with low water velocities (<15-17 cm/s) and extensive cover (e.g., shallow, quiet areas along the streambanks, backwaters, pools, beaver ponds and side channels). Jakober (1995) observed that bull trout in two streams in the Bitterroot River system moved downstream into beaver ponds and pools with complex large woody debris as water temperatures declined in autumn. Heggenes et al. (1993), who studied the winter ecology of brown trout in streams, describe this change in behavior and habitat selection as an "ecologically adaptive homeostatic response." That is, trout respond to adverse conditions (e.g., colder water temperatures and ice) by selecting habitats that minimize energy loss. This adaptation increases the overwinter survival of trout because at cold temperatures trout cannot assimilate energy as quickly and efficiently as they do at warmer temperatures (Brett 1964; Brett and Glass 1973). The speed of digestive enzymatic action at low temperatures limits metabolism, especially the rate and efficiency of food digestion and assimilation. It follows, then, that even if the digestive tract were full, limited energy would be available for metabolism in the winter. Contor (1989) notes that when temperatures drop below 4°C, trout that increase activity beyond "resting" must break down body tissue for supplemental energy even if they maintain a full digestive tract. This breakdown of tissue results in decreased condition (weight per unit length) and survival (Smith and Griffith 1994). Thus, during winter conditions, trout will survive at higher rates in quiet areas, where little to no energy is spent in swimming against the stream current. Below we describe the habitats used by trout in ice-covered

streams.

## 3.2 WINTER HABITAT CHARACTERISTICS

Following the working hypothesis that trout choose winter habitat to minimize energy loss, Cunjak (1996) proposed a list of criteria that define the winter habitat selected by trout in streams. He lists three criteria in order of relative importance: (1) protection from adverse physio-chemical conditions (e.g., ice, low oxygen) and access to refugia, (2) protection from predators, and (3) access to food. Cunjak (1996) described habitats that meet these criteria. For example, deep water, instream cover (cobble/boulder substrate, woody debris, and undercut banks), floodplain habitat and side-channels, and groundwater discharge zones are selected by trout during winter to avoid adverse physico-chemical conditions. Suitable depths and instream cover also protect trout from predators, while selection of low-velocity microhabitats and nighttime foraging provide access to food. Feeding at night also reduces predation risk.

### 3.2.1 Daytime Winter Habitat

Winter habitat selected by trout during the daytime is related to fish size and the availability of suitable habitat. Salmonids are typically photonegative (avoid light) during winter (Cunjak 1996). Small trout (<15-25 cm) prefer to secrete themselves in interstitial spaces in the substrate during daylight where rock diameter is directly proportional to the size of the fish (Cunjak 1988; Heggenes et al. 1993). These trout may move 15-30 cm below the substrate surface (Everest 1969; Cunjak 1996). In the Bitterroot River basin, small (<20 cm) bull trout and cutthroat trout conceal themselves in the interstices of large substrate and complex woody debris accumulations during winter days (Jakober 1995). Goetz (1997) also reported finding small bull trout concealed in cover during winter days in Cascade Mountain streams. In a small Idaho stream, Hillman et al. (1987) found that salmonids, if given a choice, prefer to conceal themselves in cobbles and small boulders along the stream edge during daytime in winter. Hillman et al. (1987) found few fish concealed in submerged vegetation or in clean substrate[4] near stream center if clean substrate was available near the stream edge. Contor (1989), Riehle and Griffith (1993), and Griffith and Smith (1993) also found that juvenile trout concentrated along shallow stream margins where they concealed themselves during the daytime among boulders with low embeddedness, in vegetation, and beneath undercut banks. In the Credit River, Ontario, however, brook and brown trout occupied stations beneath woody debris, undercut banks, and shelf ice, or within macrophyte beds during daytime (Cunjak 1996). In the Metolius River basin, juvenile bull trout associated with woody debris and undercut banks during the winter (Goetz 1991). Armstrong and Elliott (1972) observed juvenile Dolly Varden hiding in

[4]Clean substrate refers to the lack of fine sediments (silt and sand) that fill spaces between cobbles and boulders.

dense mats of debris or in areas with groundwater upwelling. Macrophyte beds may be less desirable daytime winter habitat than spaces in woody debris or substrate. For example, Griffith and Smith (1995) observed that the density of young trout in macrophyte beds declined steadily between November and January despite continued availability of the vegetation cover. Instead, most trout overwintered in any available cobble and boulder substrate.

Trout that are too large to hide in the substrate or woody debris tend to aggregate in large schools (up to 600 fish) in pools during the daytime (Bjornn and Reiser 1991). This is mostly true of larger trout (>15-25 cm) because interstitial spaces large enough to conceal them are scarce in most streams; however, if spaces are available, they will use them (T. Hillman, personal observation). In the Bitterroot River basin, bull trout and cutthroat trout larger than 20 cm aggregated in deep, quiet pools during winter (Jakober 1995). Others (e.g., Craig and Bruce 1982; Stewart et al. 1982; McPhail and Baxter 1996) have also observed bull trout in deep pools during winter. Water depths selected by trout during winter vary with availability. For example, Cunjak and Power (1986) observed that juvenile and adult brook trout and brown trout in an Ontario river selected water depths that ranged from 42.4 to 95.4 cm and 50 to 75 cm, respectively. For both species, water velocities selected by the trout were usually less than 17 cm/s. On the other hand, Griffith (1991) found that radio-tagged trout in an Idaho river selected stations in quiet water that ranged from 150 to 200 cm deep. The presence of pools, however, does not necessarily imply suitable winter habitat. For example, Cunjak and Caissie (1994) found that the accumulation of ice in a pool in the Miramichi River filled more than 75% of the pool volume between December and March, and reached the pool bottom in the deepest part. The remaining pool space was considered to be of marginal suitability because of the consequent higher water velocities caused by the ice mass.

Backwater habitats and beaver ponds are frequently used by some trout during the winter, apparently because they contain deep water and low velocities. As water temperatures declined below about 7°C in two streams in the Bitterroot River basin, numbers of bull trout and cutthroat trout declined in all habitat types except in beaver ponds, where large aggregations (80 to 120 trout) of both species overwintered (Jakober 1995). In a Wyoming stream, Chisholm et al. (1987) found that brook trout moved into a beaver pond in October and remained there all winter. In October, when water temperatures declined below 8°C, Cunjak (1996) observed more than 60 brook trout in a beaver pond in Catamaran Brook, more trout than he had observed in any other pool in the system during the previous three years. In shallow streams where ice cover is often in contact with the substrate, beaver ponds may represent the few suitable wintering sites available, so long as trout have adequate access to the ponds and water quality is not deleteriously affected (Komadina-Douthwright 1994).

### 3.2.2 Nighttime Winter Habitat

Although trout prefer to conceal themselves during the daytime in winter, many emerge from cover at night. Sexauer (1994) examined the nighttime winter habitat use by juvenile bull trout in a

Washington stream. She found that juvenile bull trout during winter nights occupied stations in low velocity pocket pools and backwater eddies. These fish were found closer to cover (woody debris) in winter than during summer. Jakober (1995) observed bull trout and cutthroat trout of all sizes move from daytime concealment cover to more exposed positions at night during winter. These fish remained active on or above the substrate throughout the winter at night. Goetz (1997) also observed bull trout emerge from daytime cover at night during winter in Cascade Mountain streams. For three consecutive winters, Hillman et al. (1989) studied the habitat and behavior of salmon and trout in the Wenatchee River, Washington. Although they observed no salmonids during the daytime in winter (fish concealed themselves in the substrate near the stream margin), they found relatively high numbers at night. These fish emerged about 30-60 minutes after dark and occupied stations on or near the substrate where water velocities were less than 5 cm/s. The fish segregated based on size, with the largest fish in the deepest water at night. These observations comport with those of Campbell and Neuner (1985), who found that juvenile and adult rainbow trout were not visible during the day in winter, but occupied inshore areas in shallow, quiet water at night. Contor (1989) studied the nighttime winter habitat use by rainbow trout in an Idaho river and also found that they remained hidden during the daytime, but emerged at night. These fish occupied nighttime stations near the stream banks where water velocities and depths were <15 cm/s and 20 to 45 cm, respectively. Contor and Griffith (1995) found that the number of trout emerging from cover at night was related to light intensity. Trout densities at night were lowest during moonlight phases or when artificial light illuminated the stream. Heggenes et al. (1993) found that brown trout emerged from cover at night during the winter and held positions just above or on the substrate. They note that the trout preferred the lowest velocity areas (0-5 cm/s), such as pools, riffle edges close to stream banks, and backwaters. They also note that the trout were more active at night throughout the winter.

Not all trout emerge from concealment cover each night; some remain hidden throughout the night regardless of light intensity. For example, Griffith and Smith (1993) estimated that only 61-66% of the juvenile trout overwintering in an Idaho river emerged each night from substrate concealment. Those that emerge appear to feed throughout the night (Contor 1989; Heggenes et al. 1993). Griffith (1991) reports that food in the form of drifting aquatic insects (especially midges) is more abundant at night in winter. The trout he collected throughout the winter were actively feeding even at a water temperature of 1°C. Thus, it appears that trout emerge under the cover of darkness to feed on aquatic insects during winter. Contor (1989), Griffith (1991), Heggenes et al. (1993), and Cunjak (1996) opine that because of reduced swimming performance at cold temperatures, trout are more vulnerable to warm-blooded predators. By emerging to feed at night, trout minimize the risk of predation and also acquire as much food as they can digest. Because the rate of digestion and assimilation is reduced at cold temperatures, feeding bouts may occur several days apart. Thus, trout that fed recently may remain concealed during the night.

## 3.3 OVERWINTER MORTALITY

Cunjak (1996) recently reviewed most of the literature on the winter habitat requirements of trout in ice-covered streams and concluded that the occurrence of large aggregations of trout in quiet water areas (e.g., pools, backwater habitats, beaver ponds) strongly indicates that winter habitat limits trout production in ice-covered streams. It is not surprising, therefore, that the overwinter mortality of trout is high in many streams. For example, Smith and Griffith (1994) summarized population studies of wild salmonids in streams that remained near 0°C for prolonged periods and were not affected by winter floods. In combination, these studies incorporated a total of 24 population estimates, and the overall average fish mortality rate during the first winter of life was about 50%. Needham et al. (1945) reported that on average 62% (range 16-85%) of age-0 and 80% (range 48-91%) of larger (>10 cm) brown trout died during four winters in Convict Creek, California. Maciolek and Needham (1952) estimated that 50% of the trout they marked died over a mild winter in experimental channels in Convict Creek. Cerven (1973) reports that 97% of all ages of brown trout, 73% of all ages of cutthroat trout, and 45% of stocked rainbow trout died during winter in the Temple Fork of the Logan River, Utah. Winter mortalities of rainbow trout in South Willow Creek, Montana, averaged 32% during two winters (Schrader 1989). In studies of brook trout over 11 years in Lawrence Creek, Wisconsin, Hunt (1969) noted winter mortalities that averaged 46% (range 27-65%). Lawrence Creek probably produces fish with better winter survival than ice-covered streams because it has warmer water temperatures and no anchor ice.

Physical damage from snow and ice, and fluctuating flows associated with ice formation and dispersal, appear to be the major causes of trout mortality in ice-covered streams with natural flow regimes (Needham and Slater 1944; Needham et al. 1945; Maciolek and Needham 1952; Reimers 1957; Needham and Jones 1959). Trout mortality during winter has been observed because of: (1) crushing or suffocation from collapsing snowbanks and ice (Needham and Slater 1944; Needham et al. 1945), (2) stranding from flow fluctuations caused by anchor ice formation and dispersal (Maciolek and Needham 1952), and (3) asphyxiation from ice crystals plugging gill lamellae (Tack 1938). In Temple Fork, Cerven (1973) found that anchor ice was significantly and positively correlated with juvenile brown trout mortality. Smith and Griffith (1994) studied the relationship between juvenile trout mortality and suitable winter cover (cobble substrate) and water temperature. They found that mortality of juvenile rainbow trout in cages ranged from 0 to 37%. They noted that trout survival was 11 to 24% higher in cages with cover than in cages without cover. The highest mortalities occurred in cages where water temperatures were near 0°C. Virtually no mortality occurred in cages placed near springs where water temperatures remained near 7°C. Smith and Griffith (1994) state that mortalities of trout in cages were probably lower than those of free-living trout because the cages protected fish from shifting ice and predators.

It is apparent that winter is a very critical period for trout in ice-covered streams. Even in streams with suitable winter cover and natural flow regimes, mortalities can be high because of icing

conditions. As we discuss next, stream ice can rapidly change the hydraulics and habitat of streams. These natural changes in streamflows affect the habitat use and survival of trout.

## 3.4 EFFECTS OF STREAM ICE ON TROUT HABITAT

In this section we describe the types of ice that form in streams and the impacts of ice on trout habitat. Most of the information on ice formation comes from Ashton (1986), Prowse and Gridley (1993), and Prowse (1994).

### 3.4.1 Ice Formations

Ice is produced in streams when water reaches the freezing point, provided the stream area has a quiet surface. Moving water (riffles) apparently does not crystallize at the freezing point, but will form frazil ice if it is slightly super-cooled (i.e., the temperature of the water drops below the freezing point). Whether a stream at a given location reaches the freezing point depends on its balance of heat input and loss. Heat is added by friction with the substrate, by conduction from warmer surfaces, by thermal (infrared) radiation, or by addition of warmer water. Heat is lost through conduction to colder masses, by thermal radiation, by evaporation from the surface, or by addition of colder water (including snow). Changes in ice, and consequently also hydraulic conditions, occur mostly during the night, while conditions are comparatively more stable during daytime. This periodicity in ice formation results from increased heat radiation loss during the night (Ashton 1986).

The type of ice that forms in a stream depends almost entirely on water velocity, although the nature of accumulations is a complex function of rates and durations of heat loss. Areas that are slow or lack turbulence (pools and ponds) develop surface ice; fast or turbulent areas (riffles) produce frazil ice (and consequently anchor ice) at first, then surface ice may form as ice dams reduce velocity and turbulence. Thus, there are two major types of ice that form in streams: subsurface (frazil and anchor) and surface (surface, shelf, sheet, and contour) ice. As we discuss later, subsurface ice is the most detrimental to trout.

> **Frazil Ice.**--Frazil ice is created when nuclei form in moving water (riffles) that has been cooled to a few hundredths of a degree below the freezing point. Production of these small particles continues as long as heat loss is fast enough to maintain slight super-cooling. Once formed, frazil crystals grow into discs up to several hundredths of an inch in diameter, and are very adhesive to each other and to any rough objects in the channel (e.g., rocks, branches, debris, etc.). Even though heat released by formation of the first cloud of frazil crystals can stop crystal condensation, the nuclei already produced can enlarge if heat is removed at the freezing point.

**Anchor Ice.**--Anchor ice is composed of frazil discs that have been carried to the bottom by turbulence and have adhered to rough surfaces or other frazil crystals. As long as water temperature remains at freezing and heat is escaping, anchor ice accumulations can grow even in extremely high velocities. Consequently, anchor ice readily forms obstructions that can dam streams into a series of "stairstep" pools.

**Surface Ice.**--Surface ice can form either by growth of crystalline sheet ice in quiet water, or by accumulation of floating frazil discs or "flocs" (agglomerations of discs) laterally from the shore or other emergent objects. Frazil ice can float to the surface and wash into quiet pools or eddies. Both sheet and floating-frazil surface ice can become very thick, but the type formed from frazil can thicken more rapidly by accumulating ice freed from upstream areas. Drifting frazil ice or drowned snow crystals can also adhere to the undersurface of sheet ice, and rapidly create "underhanging dams" that constrict flow in pool areas.

**Shelf Ice.**--Shelf ice is surface ice that accumulates above normal water level as instream ice production increases water surface level by obstructing flow. When the ice obstructions melt, surface ice remains suspended over the lowered stream surface. Depending on the thickness of the ice and its physical support from banks or emergent objects, shelf ice consists of anything from narrow shelves along the shore to complete caps that persist for weeks or months. The longer and more severe a period of low temperature, the more ice accumulates at the surface and the thicker and more complete will be the shelf.

**Sheet Ice.**--Sheet ice forms when water is quiet (as along stream banks) or non-turbulent. The ice nuclei that form grow into networks. Sheet ice in streams is structurally the same as that forming in ponds and lakes, and can be recognized by its smooth surface and transparency. It is usually thin, so it often collapses to form "slides" along the banks.

**Contour Ice.**--Contour ice consists of slightly elevated shelves that follow surface contours of the most precipitously descending stream sections. It forms only when air temperatures are below freezing but water temperature remains above the freezing point. Contour ice can grow laterally from any emergent object, such as a bank, root, boulder, or even an ice-shelf remnant. If the water surface continually oscillates, as in a riffle, it wets emergent objects for a few milliseconds, then withdraws and allows the water to freeze and accrete. A plate of ice at the highest level of oscillation thereby grows from its support, and can eventually join with plates from other supports to cap a stream. Contour ice is dense and clear, and can suspend more weight for its thickness than other varieties of ice.

### 3.4.2 Effects of Ice on Trout

Ice and stream-ice processes have profound effects on the winter habitat of trout and their overwinter survival in streams (Power et al. 1993). Anchor and frazil ice (subsurface ice) are considered to be the most detrimental to fish (Hynes 1970). Brown et al. (1994) reports that frazil ice poses a serious threat to trout living in high gradient streams both through direct physiological effects and by causing rapid and profound changes in habitat. They noted that as frazil crystals form, they directly affect the respiratory system of trout. When the ice crystals are small, they abrade the gills and cause hemorrhaging. As the crystals grow and aggregate, they plug the gill rakers and eventually suffocate the trout. Tack (1938) also found that frazil ice suffocated trout during cold winter nights. Brown et al. (1994) note that fry and juvenile trout are more vulnerable to suffocation by frazil ice because their mouths are smaller and more easily plugged with ice crystals than those of larger trout.

Not only does ice affect the physiology of trout, but it also has a profound affect on their habitat. Frazil ice has a negative effect on trout habitat by aggregating on woody debris and on substrate (forms of cover used by trout during winter) to form anchor ice (Brown et al. 1994). In fact, areas preferred by trout for spawning and winter rearing (shallow gravel-cobble reaches) actually promote anchor ice formation. This type of ice excludes trout from suitable winter habitat, and worse, traps them in the habitat where they later die from suffocation, freezing, or crushing. Chisholm et al. (1987) observed habitat exclusion by ice for brook trout overwintering in steams where abundant frazil and anchor ice occurred in the riffles. Brown et al. (1994) found that frazil and anchor ice actually excluded trout from overwintering habitat that had large amounts of cover in the form of woody debris. Trout mortality increased because they were forced to use sites with little or no cover. Cunjak and Caissie (1994) found that frazil ice accumulations precluded use of pools by overwintering salmon and trout. They noted that the accumulation of ice (underhanging ice dam) filled more than 75% of the volume of a pool, and was in contact with the pool bottom in the deepest part. This greatly reduced the suitability of the habitat because of the high water velocities deflected by the ice mass (Cunjak and Caissie 1994). Brown et al. (1994) found that the presence of woody debris in pools increased the buildup of underhanging dams. Thus, these pools, which are preferred winter habitat for trout, are quickly reduced to unsuitable habitat as frazil ice decreases rearing space and cover and increases velocities.

Power et al. (1993) note that substrate scouring, dewatering of stream sections, and freezing of redds are common causes of mortality that are attributed to anchor and frazil ice. Brown et al. (1994) report that incubating trout eggs and alevins of fall spawning species (e.g., bull, brown, and brook trout) are damaged or displaced by anchor ice. During the formation of anchor ice, eggs and alevins in redds are destroyed or displaced when the ice freezes the stream bottom. As the anchor ice breaks up and lifts from the bottom, it detaches substrate materials and exposes eggs and alevins. The continued build up and break up of ice dams dewaters redds and scours the stream bed crushing eggs and alevins in the process (Calkins 1989).

Not only do ice dams destroy eggs and alevins in redds, but they also dislodge or displace older trout from winter habitat when the dams break and scour the stream bottom (Power et al. 1993). Large fluctuations in water depths and velocities occur as ice dams form and break up. As ice dams build, the stream surface elevates (increased water depth and decreased velocities) upstream from the dam because the stream is forced out of its bed. Downstream from the dam, water depths decrease resulting in dewatering of some trout habitat. Trout in these locations are trapped in dewatered areas and die from exposure to freezing air temperatures. Fish upstream from the ice dams move into the impoundments and conceal themselves in the substrate or debris along the margins of the impoundments. Larger fish that cannot conceal themselves in the substrate or debris may aggregate in the deep, quiet portions of the impoundments. If sufficient heat is absorbed during the day, the dams break and the stream quickly returns to its former bed (rapid increase in water velocity and decreased depth), displacing fish that aggregated in the pools and stranding and killing those concealed in the overflow areas. Maciolek and Needham (1952) and Needham and Jones (1959) reported on the mortalities of trout, by suffocation, after ice dams broke and dewatered stream side-channels and braids. They note that trout moved into these areas as water depths increased behind ice dams and then became stranded and died when water levels subsequently receded after the ice detached from the substrate.

## 3.5 EFFECTS OF TIMBER HARVEST ON WINTER HABITAT

Because reduced metabolic demands of trout at low water temperatures lessen or eliminate time spent defending territories and feeding (i.e., summer activities), suitable habitat is the primary factor that regulates trout populations in winter (Chapman 1966; Mason 1976). Therefore, any land-use activity that affects the amount of suitable winter habitat in a stream will also have some effect on the trout population. Cunjak (1996) reviewed the impacts of land-use activities on winter habitat of fish. He notes that water withdrawal and its direct influence on reducing available habitat (by increasing icing conditions and decreasing water volume) probably affects fish populations more than any other winter alteration of streams. To a lesser extent, improper forestry activities can affect winter habitat for trout. For example, improper forestry activities may introduce fine sediments to streams, especially where riparian vegetation is removed (Hartman and Scrivener 1990). This can reduce the suitability of winter habitat for small trout that overwinter beneath medium to large substrate (i.e., cobble-boulders). In addition, improper forestry activities can reduce or damage pools, side-channels, backwaters, large organic debris complexes, and some undercut stream margins that provide overwintering trout low water velocity refugia and instream cover. Removal of riparian vegetation may also lower winter stream temperatures (Chamberlin et al. 1991), thus possibly increasing ice formation in the stream.

# 4.0 ROLE OF SMALL STREAMS IN MAINTAINING TEMPERATURES IN DOWNSTREAM FISH-BEARING WATERS

Most of the stream channels within a given drainage consist of low-order streams (Strahler 1957; Beschta and Platts 1986). Within the Swan River basin, for example, there are 1,257 miles of intermittent and perennial stream channels, of which 689 miles are first-order channels with an average gradient of 19.3%. Within the Thompson River basin, there are 1,326 miles of intermittent and perennial stream channels, of which 832 miles are first-order channels with an average gradient of 20.3% (Jensen and Dean 1996, 1997).

Most salmonid spawning and rearing takes place in the second- to fourth-order streams (Chamberlin et al. 1991). These streams tend to be larger and have lower gradients than first-order streams typically have. Within the project area, fish presence-absence surveys have shown that 95% of the reaches that contained salmonid fish had gradients ≤ 10%, and over 80% of fish-bearing reaches were ≤4% gradient (Light et al. 1998). For the Swan and Thompson Rivers, this equates to streams of second order and higher.

As current riparian rules for forest practices have developed (see Section 5.1), many small, perennial fish-bearing streams have been included in riparian leave rules in order to protect instream habitat and resident fish (first and second-order channels). An additional, or potential, concern could be identified for small perennial non-fish-bearing streams (typically, first-order channels). If a stream crosses a harvest unit and has significant shading removed, the potential exists for stream temperatures to increase and for the stream to deliver warmer water to a fish-bearing receiving stream. This section examines the effects of small, non-fish-bearing perennial streams on water temperatures in downstream fish-bearing waters.

## 4.1 BACKGROUND

The principal source of heat for small mountain streams is solar radiation striking the surface of the stream (Brown 1969). The amount of sunlight reaching the stream depends on the stream surface area and the shading provided by vegetation and topography (Brown and Krygier 1970). Vegetation typically provides substantial shade to streams in forested areas, and canopy closure measurements closely approximate overall shading levels. The magnitude of potential temperature change with streamside vegetation varies with stream size (Adams and Sullivan 1990).

Stream temperatures are very reach-specific, as equilibrium is sought in response to reach-level parameters such as shading and groundwater inflow rate. Thus, the potential exists for stream temperatures to increase or decrease in response to local conditions. Smaller streams will both heat

and cool more quickly, in response to changes in environmental parameters, than larger streams. For a full discussion of the physics of stream heating and cooling, see Sullivan et al. (1991).

Several environmental parameters can affect this situation. Lower-order streams will tend to have a larger component of groundwater input to flow than larger streams and thus tend to be cooler than higher-order streams. However, smaller streams cool and warm faster than larger streams, which will counteract the cooling trend from groundwater. A small stream also does not have the ability to affect the temperatures of a larger stream unless it contributes a fairly significant component of the resulting flow. This is demonstrated using a simple mixing equation (Brown, 1969). Given that $T_1$ and $Q_1$ are the temperature and flow of the small stream, and $T_2$ and $Q_2$ are the temperature and flow of the receiving stream, the temperature of the combined stream downstream of their confluence is expressed as:

$$\frac{(T_1 * Q_1) + (T_2 * Q_2)}{(Q_1 + Q_2)}$$

This relationship is presented in Figure 4.1, assuming tributary temperatures of 14°C, 16°C, and 18°C, and a receiving stream temperature of 12°C. The small stream will have an increasing ability to change the temperatures at the confluence of the two streams as the flow in the small stream increases. For instance, a 14°C tributary, contributing 35% of the combined flow, will increase the downstream temperature by 0.7°C. An 18°C tributary will increase downstream temperatures 1.2°C at 20% of the combined flow.

## 4.2 LITERATURE REVIEW

Research on temperature regimes of small non-fish-bearing streams is not extensive. Results from monitoring of small forested, non-fish-bearing streams in Western Washington, and their receiving waters, showed that small streams reacted in a similar manner as larger streams. Environmental parameters such as shading, air temperatures, groundwater temperatures and groundwater proportion of flow were all important factors influencing temperatures in small streams. Effects from groundwater appeared to be localized and to vary between sites. Small streams responded quickly to local conditions. Stream temperatures equilibrated to increased shading conditions (both brush-level and tree shading) over distances of about 500 feet (which equates to travel times on the order of one to two hours). Most small streams in forested drainages were tributary to streams much larger than themselves, and it was uncommon for a small stream to reach even 10% of the combined flow. Small tributaries overall had very little effect on temperatures of downstream waters, usually because of their small amount of flow in proportion to the combined flow downstream (Caldwell et al. 1991).

Research documenting increased in temperatures in fish-bearing forested streams after shade removal is fairly widespread. Research prior to 1987 is summarized by Beschta and others (1987).

Temperature decreases with additional shading levels in small forested streams, as streams move from an open area into a shaded area have also been documented. In an old-growth Douglas-fir stand in the Cascades, a temperature decrease of 4.5 °C was observed after water flowed from an unshaded reach through 700 feet of shaded channel (Levno and Rothacher 1967). In a small Appalachian stream, temperatures decreased 6.6 °C after the stream had crossed 400 feet of channel shaded by Douglas-fir (Swift and Messer 1971). In England, temperature decreases of up to 5 °C were noted within less than 1000 feet of woodland shade (Gray and Edginton 1969, in Rishel and others 1982). Since small streams respond faster to changes in shading than larger streams, we would assume that small non-fish-bearing streams would cool down in the shorter range of the distances if shading was present.

## 4.3 DISCUSSION

Because stream temperatures are determined by the environmental conditions at a reach level, and because, in many cases, small tributaries are too small in size to affect temperatures in the receiving stream, the potential for small non-fish-bearing streams to affect temperatures downstream exists in specific situations only.

Because of the relative size that a non-fish-bearing tributary needs to be in relation to the downstream reach, there are two situations where small tributaries have the potential to deliver warmer water to downstream fish-bearing waters. The first is where a non-fish-bearing tributary contributes a substantial portion of the combined flow in the fish-bearing stream. (The actual proportion will depend on the temperatures of each, although using the mixing equation above, it appears that the tributary needs to be on the order of 20% of the combined flow.) The second situation would be where two non-fish-bearing tributaries combine to form a fish-bearing stream. In both these cases, if shading were to be removed from the lower reaches of the non-fish-bearing tributaries, the potential exists for warmer water to be delivered downstream.

Because very small streams heat and cool more quickly than larger streams, a solution to these potential problems can be developed. Many small streams are potentially shadeable by brush-level vegetation as well as by trees. The addition of shading to the small tributary just upstream of the fish-bearing water for an effective distance will control water temperatures at the confluence with fish-bearing waters. The specific distance will depend on the temperature standard, or level of acceptable temperature change with shade removal, specified in each state's regulations. For example, equilibration distances of 500 feet or less were measured in Western Washington for streams flowing from unshaded intro shaded areas (Caldwell and others 1991). This observation is similar to distances from the literature for larger forested streams presented above.

Another concern that is expressed with respect to small streams warming fish-bearing streams is whether the potential exists for several small tributaries to add warmer water to a fish-bearing

stream, for a combined affect. This case was investigated for Western Washington and found that, for the ten USGS quadrangle maps of forested areas sampled, the potential did not exist. The average distance between non-fish-bearing tributaries was 1,778 feet (with a range of 650-3,320 feet) (Caldwell and others 1991). So even if the tributaries were large enough to affect the fish-bearing stream (which most would not be as stream size increases in a downstream direction) stream conditions in the fish-bearing waters would be expected to be in equilibrium with channel conditions within 500 feet, before the next tributary inflow point.

Where standard forest practice regulations retained insufficient levels of shading, a "thermal management zone" on the order of 500 feet could be established in the lower reaches of non-fish-bearing tributaries that are large enough to have the potential to affect downstream fish-bearing reaches. The most common cases would be single-channel reaches changing to fish-bearing, the confluence of two non-fish-bearing tributaries, and the inflow of a non-fish-bearing tributary constituting greater than 20% of the combined flow. The relatively lower drainage density in the project area compared to Western Washington suggests that there is little or no potential for multiple small non-fish-bearing tributaries to affect downstream fish-bearing waters in combination.

# 5.0 EFFECT OF TIMBER HARVEST ON CANOPY COVER OVER STREAMS

As discussed in Sections 4 and 6, canopy cover over streams is a major factor influencing water temperature. It is also known that forest management has the potential to alter canopy cover over streams when riparian timber is harvested. However, no studies have been conducted to examine what effect riparian timber harvesting (in compliance with current forest practice regulations in Montana and Idaho), has on canopy cover over streams. The purpose of this section is to describe an initial study that was undertaken on Plum Creek Timber Company lands in Montana and Idaho to examine what effect riparian timber harvest has on canopy cover over streams.

## 5.1 CURRENT FOREST PRACTICES REGULATIONS

### 5.1.1 Montana

In Montana, Steamside Management Zone (SMZ) regulations do not explicitly require canopy cover (or shade) to be maintained over streams. However, canopy cover is retained as a by-product of leave-tree requirements. Montana has three stream classes that dictate the level of riparian protection. Class I stream segments are defined as either 1) supporting fish, or 2) flowing more than 6 months out of the year and contributing surface flow to a downstream water. Class II stream segments are defined as either 1) having surface flow more than six months out of the year (but not supporting fish), and not delivering to a downstream water; or 2) having flow less than six months out of the year and delivering to a downstream water. Class III stream segments have flow less than six months out of the year and rarely contribute surface flow to downstream waters. For additional discussion of riparian regulations in the planning area, see Technical Report #6 (Sugden and Light 1998).

The base Montana SMZ extends 50 feet from the high water mark on each side of streams. This base SMZ extends to encompass adjacent wetlands. It also extends up to 100 feet wide where sideslopes in first 50 feet of the SMZ exceed 35%.

Current regulations for Class I and II streams require that no more than 50% of trees be harvested in the SMZ, and that leave trees must be representative of size and species of the pre-harvest stand. Bank edge trees must also be favored for retention. Though 50% of trees can be harvested, at least 87 trees per acre must be retained in Class I SMZs, and 44 trees per acre must be retained in Class II SMZ's.

With regard to influencing maximum stream temperatures, Class I streams are the streams that are at issue, since Class II intermittent (or disconnected) streams presumably do not contribute to

summertime temperatures in downstream fish-bearing waters. No studies have been conducted to date on what benefit leave-tree requirements have on maintaining canopy cover over streams or water temperatures.

### 5.1.2 Idaho

The Idaho Forest Practices Act specifically requires shade to be maintained over Class I streams. Class I streams are defined as having fish. The Act specifically requires that 75% of "existing shade" must be retained during harvest along Class I streams. In other words, if stream shading is currently 75%, a planned harvest could reduce shading levels to 56% (0.75 x 75% = 56%). This requirement is independent of tree retention requirements. In some cases, Plum Creek Timber Company foresters find that the shade requirement is the limiting constraint on riparian harvesting, while in other cases tree retention requirements are the limiting constraint (Personal Communication, John Quigley, Plum Creek Timberlands Superintendent). Though a 75% reduction of existing shade is the theoretical maximum level allowed under law, it is unknown to what extent actual timber harvest operations effect shading over streams.

Class II streams are defined in Idaho as streams that support "few if any fish." This could include perennial and intermittent streams. The Act has no specific requirements for maintaining shade over these streams. However, at least 203 trees per 1000 feet of channel (size 3"-7.9" DBH) must be retained within 30 feet on each side of the stream.

In addition to these requirements, perennial Class II streams have been further protected in the Little North Fork Clearwater River and portions of the Lochsa River through Idaho's antidegradation process. For each of these areas, Site Specific Best Management Practices were developed by a citizen committee to address the unique hazards and resources in these designated "Stream Segments of Concern (SSOC)." The Little North Fork Clearwater River SSOC encompasses all of Plum Creek's ownership in the basin. The SSOC listed in the Lochsa River basin include Boulder Creek, Brushy Fork, Crooked Fork, Spruce Creek, Walton Creek, and White Sands Creek. The only Plum Creek ownership in the Lochsa not affected by these listed streams are lands that drain to the Lochsa River below the Crooked Fork/White Sands/Walton Creek confluence. Along perennial Class II stream segments in these areas 75% of "existing shade" must be retained after harvest. These stream segments also have the same tree retention requirements as Class I streams.

### 5.1.3 Washington

The Washington Forest Practices Act contains specific provisions for maintaining canopy cover over fish-bearing streams. Requirements are based on a temperature nomograph similar to the one developed in Section 6. These canopy cover requirements are specifically designed to ensure that Washington's numeric water quality standards of 16°C for Class AA streams and 18°C for Class

A streams are met (measured as instantaneous temperatures). Most of the streams draining Plum Creek Timber Company lands in Washington are classified as AA.

In addition, canopy cover requirements along the lower reaches of perennial non-fish-bearing streams (Type 4 streams) are often developed during Watershed Analysis, with the intention of maintaining temperature levels in downstream fish-bearing waters (See discussion in Section 4).

Because of a wide-spread level of incorrect identification of small fish-bearing streams as non-fish bearing ("Type 4") streams, an emergency rule is currently in effect. This rule requires that, in the absence of site-specific information, streams of specific sizes, gradients and drainage areas are treated as fish-bearing. These streams are then subject to the canopy closure requirements for Type 3 streams.

The current emergency rule is in place until March 12, 1998. The Dept. of Natural Resources expects the emergency rule to be renewed, and to be in place for at least 18 months (until June 1999). This is due to ongoing negotiations, and the time necessary for newly proposed rules, when they are developed, to go through the hearing process (N. Sturhan, WDNR, pers. comm. 1998).

## 5.2 STUDY AREA

To be included in the study of harvesting effects on streamside canopy cover, timber harvests had to meet the following criteria: 1) Harvest would be along a perennial stream in the summer of 1997 in Montana or Idaho; 2) The harvest area must encompass at least 600 feet of stream channel, with harvest occurring on both sides of the stream; 3) No roads were to be located within the SMZ in the sample reach; 4) Timber harvest operations must be initiated after understory foliage had leafed out, and the harvest must be completed before senescence; and 5) The riparian area must be harvested to a level that came close to meeting the intent of the applicable regulations. In other words, if few or no trees were to be harvested in the SMZ, they were not included in the study.

Perennial streams were selected since they are the only ones that typically contribute surface flow at times of the year when maximum temperatures are recorded. The sample reach length of 600 feet was chosen as a reasonable distance for which an average could be obtained. Harvest on both sides of the stream was required so that a maximum effect could be measured. Timber harvest timing constraints were set to avoid complications associated with leaf-out and leaf-fall from deciduous vegetation.

A total of 10 sample sites, contained within nine separate timber harvests, were identified by Plum Creek Timber Company foresters. Nine of these sites were distributed throughout northwest Montana. One site (Site #2) was located in the St. Joe River basin in northern Idaho. No other harvests met the selection criteria. Harvest timing requirements, and the requirement for harvest to

be conducted on both sides of the stream appeared to be most limiting in terms of finding additional sample sites.

## 5.3 METHODS

### 5.3.1 Data Collection

At each site, the upstream and downstream boundaries of the 600 foot study reach were delineated with special colored flagging. Flagging was also hung at 50 foot intervals (stations) along the sample reach. Canopy cover measurements were then made at each station starting at 50 feet (station 50) and ending at station 550, for a total of 11 measurements in each study reach. The beginning and ending stations were not included in order to avoid potential "edge" effects. Canopy cover measurements were made using a concave spherical canopy densiometer model-B (Lemmon 1956a and b) using protocol outlined by Platts et al. (1987). The eleven canopy cover measurements were averaged to obtain the mean canopy cover for the sample reach just prior to, and immediately following harvesting.

In addition to canopy cover data, riparian stand composition and characteristics were measured every 100 feet at transects located at stations 100-500 in the study reach. Data were gathered on tree density, average tree height and diameter, and species using a point-quarter sampling method (Smith 1977). Only trees larger than 8 inches diameter at breast height (DBH) were included because they are the minimum harvestable size under Montana SMZ rules. The point quarter data were collected from transects perpendicular to the stream at a distance of 25 feet from the high water mark on both sides of the stream for a total of 10 measurements in each study reach. It is expected that this sample location provides an average for characteristics within 50 feet of the stream.

### 5.3.2. Statistical Analysis

Our study for assessing effects of timber harvest on canopy cover over streams incorporated a mixed, two-factor (10 x 2) design with repeated measurements on the second factor. The first factor included 10 independent survey locations (i.e., independent timber sale locations), while the second factor described canopy cover before and after timber harvest. We analyzed these data with a two-factor, mixed-model, analysis of variance with one repeated factor. We then used paired t-tests to assess differences in canopy cover before and after timber harvest for each survey location. Before running analyses, we tested data for normality and circularity assumptions.

## 5.4 RESULTS

Percent canopy cover over streams were found to differ significantly before and after timber harvest

(Table 5.1). After timber harvest, mean percent canopy cover across all survey locations decreased from 67 to 62% (Table 5.1). The effects of timber harvest on canopy cover over streams also differed significantly among the survey locations (Table 5.2). In only four of the ten survey locations did canopy cover differ significantly after timber harvest (Table 5.2, Figure 5.1). At most, canopy cover decreased from 69 to 56% at Site #6. The least change occurred at Site #9 (51.36-51.32%). In one site (#10), canopy cover slightly increased after timber harvest. This could have been due to limitations on the measurement technique, or additional leaf growth that occurred between measurements. Characteristics of the stream channels surveyed are summarized in Table 5.3.

Mean stand density decreased from an average of 165 trees per acre to 117 trees per acre following harvest (Table 5.4). Average tree height decreased from 79.7 feet to 78.4 feet, while average tree diameter decreased form 15.3 inches to 14.7 inches.

## 5.5 DISCUSSION

The four sites that had a statistically significant decrease in canopy cover following harvest (Sites 3, 5, 6, and 7) also had the highest percentage of timber harvested (Table 5.4). Figure 5.2 indicates that there may be a linear relationship between the percentage of the pre-harvest stand cut and the reduction in canopy cover ($r^2$=0.63; P=0.0067).

Though significant differences were found in four of the 10 sites, it is important to remember that this study only evaluated impacts along a single 600 foot length of stream. As mentioned earlier, harvesting on both sides of the stream is a relatively infrequent occurrence. In most cases, due to harvest scheduling, logistics, and required equipment, harvest typically only occurs on one side of the stream at a time. The exact effect of harvesting on stream temperature at the watershed scale is a function of the temporal and spatial distribution of harvests, the rate of recovery of stream shading, the magnitude of any particular harvest, and other watershed-specific variables.

With regard to the efficacy of state regulations, our results found canopy cover changes ranging from 0-13 percentage points. Using the preferred regression model developed in Section 6, the average impact on stream temperature (MWMT) from harvesting in these areas would be less than 1°C.

Further research is needed to improve our understanding of the effects of timber harvesting on canopy cover and water temperature. This study could be strengthened by adding additional sites in the coming years, and exploring options for modeling watershed-scale temperatures as a function of natural and anthropogenic disturbances through space and time.

# 6.0 TEMPERATURE PREDICTION MODEL

In this section, a linear regression model is described that predicts maximum summer stream temperature as a function of elevation and canopy cover. In addition, two indices of the severity of the temperature year were evaluated in the regression analysis to account for year-to-year climatic variability. This model could serve as a tool for determining approximately how much canopy cover would be needed along a particular stream segment in order to keep the stream below some specified maximum temperature, determining what water temperatures might be expected under natural conditions, or serve as a basis for adaptive management and monitoring. This model was developed from stream temperature data in western Montana and northern Idaho. Prior to describing this model, we provide background on the physics of stream heating and cooling, and describe similar models previously developed in the Pacific Northwest.

## 6.1 BACKGROUND

Water temperatures at a given location are a function of a balance between heat inputs and heat loss. Heat inputs are driven by solar radiation, which varies daily and seasonally, and are affected by shading from riparian vegetation and topography. Heat loss is largely regulated by the difference between air and water temperatures, as well as by conduction of heat to the stream bed and by the rate of groundwater inflow (Adams and Sullivan 1990).

Research on the effect of shading removal has mostly focused on summer water temperatures (reviewed by Beschta et al. 1987). Research summarized in that report (from Alaska, British Columbia, and Oregon) showed a typical change of 3 - 7°C in daily maximum water temperatures from significant reduction in streamside shading, although the range of results was 0 - 16 °C. More change was seen in smaller streams, probably due to their rapid response time (Brown 1969). Levels of riparian shade removal reported in these studies are not typical of current practices, but illustrate maximum potential effects.

Results from a cooperative temperature study in Washington State, covering 92 sites in several ecoregions, found an average 5°C increase in maximum water temperatures as shading decreased from 75-100% to 0-25% (Sullivan et al. 1990). This is similar to the pattern of changes reported by Holtby and Newcombe (1982), who related a measure of thermal units to proportion of vegetated streambank.

There has been less research on the effects of riparian canopy removal on winter water temperatures, and its connection to habitat alteration or changes in fish behavior. Holtby (1987), summarizing changes associated with timber harvest in a Vancouver Island stream, found monthly average temperature increases in the February-April period of 1-2°C. Sullivan et al. (1990), looking at eight monitoring sites mostly in Western Washington, noted temperatures decreased with increasing

elevation, and noted a 0.5°C difference in average February temperatures between two neighboring streams with similar elevation, and aspect and different shading levels. Both Holtby and Sullivan et al. found the greatest temperature differences between shaded and unshaded sites occurred in April (i.e., early spring), and Holtby reports an effect from the additional thermal input on coho salmon fry.

Cunjak (1996) reports that the most prevalent impact on winter habitat in northern latitude streams to be reductions in streamflow, which increase ice formation and decrease available space. He also notes that removal of riparian trees can decrease the amount of shallow edge habitat, and increase deeper, scoured channel areas, which can reduce winter habitat (see Section 3.5).

Local water temperatures can change where cooler or warmer tributary waters, or groundwater, enter a stream channel. The magnitude of changes is in proportion to the temperatures and amounts of flow in each source (Brown et al. 1971).

Sullivan et al. (1991) discuss what is known about stream temperatures at the basin or watershed level. Temperatures tend to increase in stream reaches as water flows from headwaters to lower elevations (Hynes 1970; Theurer et al. 1984; Sullivan and Adams 1990). This appears to be caused by several factors: air temperature tends to increase as elevation decreases, groundwater inflow is less in proportion to in-channel flow, and wider stream channels result in decreased shading from riparian vegetation (Beschta et al. 1987).

### 6.1.1 Previous Investigations

Using data primarily from forested streams in western Washington, multiple linear regression models were developed to predict maximum water temperatures using a combination of site and stream characteristics. Several good empirical relationships between stream characteristics and water temperature were developed based on five environmental variables: stream shading, mean air temperature, elevation, stream discharge and bankfull width. Typically, a combination of local environmental factors had an important influence on water temperature, and no one factor alone was a good predictor of stream temperatures. High values for air temperature and deeper stream depths tended to increase annual maximum temperatures, and higher shading levels and rates of groundwater inflow tended to decrease annual maximum temperatures (Adams and Sullivan 1990; Sullivan et al. 1991).

An empirical model also based on the monitoring data was developed to identify stream reaches in Washington potentially sensitive to reductions in shading as a result of forest practices. The best model, using temperature, elevation and shading information, predicted which of 39 Western Washington sites might exceed water quality standards approximately 89% of the time (Sullivan et al. 1990). Since that time, additional temperature monitoring in Eastern Washington has been done,

and a similar screen for Eastern Washington has been developed based on linear regressions of maximum temperature with respect to shading and site elevation (CMER 1993).

The amount of sunlight reaching the stream depends on the stream surface area and the shading provided by vegetation and topography. Vegetation typically provides substantial shade to streams in forested areas, and canopy closure measurements closely approximate overall shading levels (Brown 1969; Brown and Krygier 1970). Therefore, we used canopy cover along with elevation in the nomograph; canopy cover to represent overall shading levels and elevation to represent the general cooling of air temperature regimes as elevations increase.

Temperature nomographs developed from monitoring data for the Western Washington and Eastern Washington ecoregions are used to estimate target levels of shading necessary for protection of water temperatures, using estimates of site elevation and current and potential stream shading levels (see Washington Forest Practices Act). These are effective parameters to use at a predictive or planning level. Air temperature tends to vary with elevation at this latitude, and shading levels control solar inputs and amounts of heat loss. Shading levels are also potentially affected by forest practices.

## 6.2 STUDY AREA

Water temperature data were collected in 100 streams in western Montana and northern Idaho between 1991 and 1997 using electronic thermographs. Data sources include Potlatch Corporation, Idaho Division of Environmental Quality, and Plum Creek Timber Company. Stream sizes varied from 2nd to 5th Order (Strahler 1957). The primary drainage basins from which these data were collected included the Blackfoot, Swan, and Thompson River basins in Montana; and the St. Joe, Lochsa, St. Maries, Palouse, and Potlatch River basins in Idaho.

## 6.3 METHODS

### 6.3.1 Data Collection

All temperature data were collected with electronic thermographs manufactured by Onset Computer Corporation (Pocasset, MA). Average canopy cover over each stream reach was measured over a distance of 1000 feet upstream of each thermograph location using a concave spherical densiometer model-B (Lemmon 1956a and b). Protocol for densiometer measurements are described by Platts et al. (1987). At least eleven canopy cover measurements were averaged over the 1000 foot upstream reach to obtain an estimate of mean canopy cover. Elevation at each thermograph site was obtained from USGS 7.5 minute topographic maps.

### 6.3.2 Statistical Approach

Multiple linear regression was used to relate physical variables to both the maximum weekly maximum temperature (MWMT) and the maximum weekly average temperature (MWAT) for data collected between 1991-1997 at 100 sites. The number of years sampled at each site varied from 1 to 7 for a total of 220 observations. The objective of this regression analysis was to provide a predictive model for estimating the maximum weekly maximum temperature (MWMT) and maximum weekly average temperature (MWAT) at a given location based on physical variables.

An initial regression analysis was conducted using elevation and canopy cover over the stream as independent physical variables. Analysis of covariance (ANCOVA) was used to determine whether these analyses should be partitioned by year since mean temperatures varied among years. Attempts to significantly improve this regression by using year specific variables were also made by including modified Palmer drought severity indices (Palmer 1965; Alley 1984) for Idaho (region 4) and Montana (region 1), and air temperatures from St. Maries, Idaho, Missoula, Montana, and Kalispell, Montana. Monthly drought indices were summed for each year for the periods January-August, March-August, May-August, June-August and July-August and assigned to sites from the appropriate state. Temperatures expressed as deviation from mean daily air temperatures for July and August for the three locations were used in the regressions. Temperatures for each site were assigned from the closest of the three weather stations.

The quality of individual regression models was evaluated in several ways. First the multiple $r^2$ of the regression was compared between regression models with higher values representing better fits. Second, the average prediction sum of squares (PSS) was calculated. PSS is the squared difference between the actual and predicted temperature for the $i^{th}$ observation using a regression equation formulated with the $i^{th}$ observation deleted. The smaller PSS, the better the predictability of the model. Average PSS was calculated as PSS divided by the sample size to standardize for differences in sample sizes between regressions. Third, the significance of regression coefficients was evaluated. Only models where all regression coefficients were significantly different from zero ($p < 0.05$) were considered. Fourth, condition indices were used as a measure of colinearity or covariance between regression variables. Models with condition numbers above 30 were considered unacceptable (Belsley et al. 1980). Finally, residuals were visually evaluated for extreme outliers and for consistent patterns.

The number of times sites were sampled varied from 1 to 7, potentially resulting in pseudo-replication problems or biases incurred with multiple data from the same site. This was not considered to be a significant problem as an individual site sampled 7 times during the seven year period contributed only 3.2% of the total samples. Additionally, these sites were sampled in different years and were subject to different drought index and temperature conditions.

## 6.3 RESULTS

### 6.3.1 Current Conditions

Elevations of sample sites ranged from 1960 feet to 5670 feet with a mean elevation of 3534 feet, while canopy cover ranged from 9% to 100% with a mean canopy cover of 53% (Table 6.1). Maximum weekly maximum temperatures (MWMT) for samples ranged from 8.0°C to 25.5°C with a mean MWMT of 14.8°C for all sites. Maximum weekly average temperatures (MWAT) ranged from 6.8°C to 22.1°C, with a mean MWAT of 12.9°C.

### 6.3.2 Statistical Analysis

ANCOVA's using canopy cover and elevation as covariates were used to test for differences in temperature among years, using data from all years (1991-1997) and years with at least 10 samples (1994-1997). Significant differences ($P < 0.000001$) in average stream temperature by year of sampling were found for both analyses. Mean stream temperature for sites sampled in 1994 (17.1°C, $n = 38$) were found to be significantly different ($P < 0.05$, Tukey-Kramer multiple comparisons) from mean MWMT for sites sampled in 1995 (14.5°C, $n = 82$), 1996 (15.1°C, $n = 38$) and 1997 (14.2 °C, $n = 45$). Numbers of sites sampled for the other years (1991-1993) were insufficient to obtain suitable mean temperatures ($n = 3$ to 7). These results suggest that regressions should be done separately for each year, or with additional parameters included to account for among-year variation.

Individual regressions for stream temperature (MWMT and MWAT) using canopy cover and elevation as independent variables were performed for each year (Tables 6.2 and 6.3) where sufficient numbers of sites were sampled (1994-1997). For MWMT, the regression for 1994 had the highest multiple $r^2$ (76.6%) and lowest average PSS (4.66) indicating the best regression for the individual years. The 1996 regression had the next largest multiple $r^2$ (56.1 %), but had the highest average PSS (7.46). The two years with the highest temperatures had the highest multiple $r^2$ suggesting that in years with high water temperatures, elevation and canopy cover had the most influence on water temperature. Combining data for all years produced a lower multiple $r^2$ (49.2%). Regressions using elevation and canopy cover individually had much lower $r^2$ values (19% to 37% using all data) and were not considered adequate predictors of MWMT or MWAT (data not presented).

Because average stream temperature varied by year, regressions using two indicators of yearly variation in stream temperature were added to the multiple regressions to generate relationships using all data. These were the modified Palmer drought severity index (DI) (Palmer 1965; Alley 1984) and July-August air temperature (AT) deviations from normal. The best four models using three variables all included elevation and canopy cover in addition to: July-August DI, June-August

DI, July-August AT, or July AT (Table 4.2). DI and temperature deviations for these models are shown in Table 4.4. The best model with three variables included elevation, canopy cover and July-August DI. Models with June-August DI, July-August AT, or July AT were nearly as good with rather similar multiple $r^2$ values (55.4 to 57.0%). The four best models with four variables had nearly identical multiple $r^2$ values (all 58.8%), and were only slightly better than the best models with three variables.

Since lower (more negative) modified Palmer DI represents dryer conditions and relates to higher stream temperatures in the regression equations, the reoccurrence frequency for values measured in 1994 (the driest year) was assessed. An analysis of the frequency distributions of DI for both Montana (region 1) and Idaho (region 4) was conducted to using modified Palmer DI for the period 1895-1997. This analysis was done for the average DI calculated for the period June-August. Table 4.5 shows the percentile values for these drought indices. The point estimate for the Montana 1994 DI (-2.27) was at approximately the 18 percentile, or reoccurring about every 5-6 years. The 95% confidence interval place the reoccurrence interval between approximately 4 and 8 years.

For the Idaho DI for 1994 (-3.60), the point estimate was at the 5 percentile with a recurrence interval of 20 years. The 95% confidence interval place the reoccurrence interval between 10 and 100 years.

## 6.4 DISCUSSION

Past research has indicated that use of shading and elevation parameters is often very effective in predicting summer maximum water temperatures (Sullivan and others 1991). We wished to explore whether the Palmer drought severity index (DI) (an indicator of antecedent meteorological conditions) and/or summer air temperatures (an indication of summer climate compared to long-term averages) could be combined with elevation and shading parameters to improve model predictions. These predictions could then be used to improve nomographs, which would allow managers to predict shading levels necessary to maintain target temperatures at varying elevations, and account for annual variation.

Table 6.2 presents the results of the development of 13 regression models for MWMT. Most models explained a similar amount of the variation in the data used to develop them, except the model using 1994 data alone. Addition of various combinations of the two climate parameters improved model performance; the drought index used alone predicted slightly more variation than did summer air temperatures used alone. Using both climate variables did not improve predictions much compared using only one or the other.

A regression model based on the full data set would allow model use over as wide a range of interpolated site elevations as possible. Our next decision was whether to include one of the two

climate parameters (drought severity index or summer air temperatures) into the model. No difference in improvement of model performance would result from the choice of one climate variable over the other, and little improvement resulted including both climate variables. The model using all years of data and incorporating the July-August drought severity index predicted fairly well Distribution of residual values was normal for both the MWMT and MWAT metrics, varying around the line of one-to-one correspondence between measured and predicted values. This is illustrated for MWMT in Figure 6.2.

We chose to include the drought severity index because it is a commonly used parameter, and is calculated on a real-time basis by the National Weather Service for all regions of the United States. Because DI values have been calculated since 1895, current climate conditions and trends can be placed in a historical context. This may allow additional analysis of the differences between observed temperatures at the same site over different years.

The preferred prediction equations are summarized as follows:

MWMT = 27.14003 - (0.001870*Elevation) - (0.09982*Canopy Cover) - (0.89125*DI)

MWAT = 24.58006 - (0.002196*Elevation) - (0.06792* Canopy Cover) - (0.82252*DI)

Where *Canopy Cover* is measured over the stream channel in the upstream reach (using protocol outlined in Platts et al. 1987), *Elevation* is measured in feet, and *DI* is the mean July-August Modified Palmer Drought Severity Index.

Assuming average drought conditions (DI=0), equations would simplify to the following:

MWMT = 27.14003 - (0.001870*Elevation) - (0.09982*Canopy Cover)

MWAT = 24.58006 - (0.002196*Elevation) - (0.06792*Canopy Cover)

Several potential uses for a series of nomographs developed from this regression model can be envisioned. Figure 6.3 is an example nomograph, using the regression model predicting MWMT temperatures under average climate conditions (DI = 0). Each line represents a target temperature, and the Y-axis indicates the level of minimum canopy closure associated with maintaining the target temperature.

This information can be used to describe the range of current conditions in the project area. Estimates of the natural, or potential, levels of temperature and canopy closure can also be made. For instance, the nomograph indicates that expected MWMT temperatures at elevations near 2000 feet will be 14°C or more, even where canopy closure levels are high.

The nomograph could also be used in a operational design context, to estimate levels of canopy closure necessary to attain target temperatures under different climate conditions (see Sullivan and others 1991). For instance, this nomograph indicates that, at 4000 feet in elevation, MWMT temperatures will approach 14°C only where canopy closures exceed 60%. Because the drought severity index can be varied to reflect a range of potential conditions (i.e., moderately wet, normal, moderate drought, severe drought), the resulting nomographs could be used to explore the different effect of canopy closure on MWMT temperatures using varying assumptions about climate conditions.

The nomograph concept also can be used to develop and test hypotheses as part of research and adaptive management. For instance, as additional information becomes available regarding the post-harvest canopy closure levels that result from the riparian prescriptions developed for this project, the nomographs could be used to estimate what changes in maximum stream temperatures within the project area would be predicted to occur.

# 7.0 SUMMARY

This report addressed several questions related to the management of riparian areas for maintenance of stream temperatures suitable for native fish in Plum Creek's Habitat Conservation Planning Area. The purpose of the report is to provide an intelligent foundation on which Habitat Conservation Plan (HCP) commitments can be developed. A summary of the reports major findings for each question posed are as follows:

1) **What are the temperature requirements of bull trout and other native salmonids in the HCP area?**

Adult bull trout were found to rear and migrate within a wide range of temperatures (4-20.5°C). This range is similar to those observed during steelhead migration (<21°C; Table 2.5). Temperatures at the time of bull trout spawning range from 4 to 12°C, which is also similar to other fall-spawning salmonids in the project area. Because mountain whitefish are late-fall spawners, they tend to spawn at colder temperatures than both chinook salmon and bull trout. Steelhead, redband trout, and westslope cutthroat trout spawn during spring at temperatures between 2 and 20°C. As with chinook and bull trout, optimal spawning temperatures for steelhead, redband trout, and westslope cutthroat trout tends to be ≤10°C. Incubation temperatures for other native salmonids in the project area are warmer than those for bull trout. Incubation temperatures for chinook salmon, a fall spawner, range from 5 to 14.4°C. In comparison, incubation temperatures for bull trout range from 1 to 6°C.

Juvenile bull trout can rear within a wide range of water temperatures (4-20.5°C); however, optimal temperatures appear to fall between 10 and 15°C. This optimal temperature range appears to be cooler than that for other native salmonids in the project area. For example, steelhead, rainbow trout, and chinook all have optimal temperatures that range from 10°C to well above 15°C. The upper optimal temperature for rainbow trout is 22°C. Juvenile steelhead avoid temperatures greater than 22°C. The optimal temperature range for cutthroat trout (12-15°C) falls within the optimal temperature range for bull trout. In addition, preferred temperatures for native salmonids in the project area are similar to the optimal temperatures of bull trout.

2) **What are the winter requirements of fish, and how can timber harvest effect their habitat during this time of year?**

The behavior, habitat use, and survival of trout at very cold temperatures (winter) differs considerably from that during warmer, ice-free periods. Much research indicates that winter conditions can limit trout production in streams. Overwinter mortality of trout can reach 97%, but generally averages about 50%. Suitable rearing space is the primary factor that

regulates stream fish populations in the winter. Choice of winter habitat is governed by the need to minimize energy expenditure, with the main criteria being protection from adverse physico-chemical conditions (e.g., ice, spates, low oxygen) and predators. Consequently, trout are typically more active at night than during daylight hours in winter. Because of reduced metabolic demands of trout during winter, trout occupy stream locations with low water velocities (<15cm/s) and extensive cover (e.g., clean substrate, large woody debris, undercut banks, deep pools, beaver ponds, and side channels). These areas used by trout during winter can be affected by improper timber harvest, which can reduce or damage pools, side channels, large woody debris complexes, and undercut banks. In addition, improper harvest can increase deposition of fine sediments in streams, which reduce winter rearing space for juvenile trout. Removing stream shade may also make streams colder during winter, thus possibly increasing ice formation in streams.

3) **Given that many streams in the planning area are non fish-bearing, to what extent can timber management along these streams influence downstream temperatures in fish-bearing waters?**

Non fish-bearing streams constitute a large fraction of the drainage network in the planning area (Jensen and Dean 1996; Jensen and Dean 1997). Available research suggests that these streams do have the potential to influence summertime water temperatures in downstream fish-bearing reaches when they are perennial, and where they add a significant percentage of the streamflow to the fish-bearing reach (typically more than 20%). In addition, due to the typically long travel times for small non fish-bearing streams, their temperatures tend to equilibrate to surrounding environmental conditions in relatively short distances (e.g., 500 feet). As such, an effective management strategy for controlling temperatures in downstream fish-bearing waters would be to ensure that sufficient shading is available in the lower 500 feet of these streams (before they enter a fish bearing stream) where the non-fish streams contribute a sufficient fraction of the streamflow (e.g., 20%).

4) **How do contemporary forest management activities (in compliance with existing regulations in Montana and Idaho) alter canopy cover (shade) over streams?**

Canopy cover and riparian stand conditions for nine streams in Western Montana and one stream in Northern Idaho were examined before and after timber harvest in 1997. Canopy cover reductions following harvest ranged from 0-13 percentage points. Mean canopy cover before harvest was 67.0%, while mean canopy cover after harvest averaged 62.4%. Of the ten sites measured, four had statistically significant decreases in canopy cover ($P<0.05$). Results also suggest that there may be a linear relationship between the percentage of the pre-harvest stand cut and the reduction in canopy cover ($r^2=0.63$; $P=0.0067$). With these levels

of canopy cover change, stream temperature changes are expected to be small (<1°C) based on the predictive model developed in Section 6.

5) **Can a practical predictive model be developed to help facilitate management questions?**

Multiple linear regression was used to relate physical and meteorologic variables to both the maximum weekly maximum temperature (MWMT) and the maximum weekly average temperature (MWAT) for data collected between 1991-1997 at 100 sites. Most regression models explained a similar amount of the variation in the data used to develop them (38-58%), except the model using 1994 data alone (77%). Addition of various combinations of the two climate parameters improved model performance; the drought index used alone predicted slightly more variation than did summer air temperatures used alone. Using both climate variables did not improve predictions much compared using only one or the other. The preferred models incorporate the entire data set and the July-August drought index.

The predictive models can be used to describe the range of current conditions in the project area, estimate maximum stream temperatures under natural (or potential) conditions, or develop and test hypotheses as part of research and adaptive management.

The information presented in this report will be used in concern with other scientific information to build a technical foundation on which a defensible habitat conservation plan for native fish can be developed. As with any science, areas of uncertainty exist. Though stream temperature has been extensively studied in the coastal Washington and Oregon for over 25 years, little research exists for the northern Rocky Mountains. With regard to temperature requirements of bull trout, laboratory investigations are just being initiated to fill critical information gaps. Plum Creek plans to address significant areas of uncertainty in our scientific knowledge with an adaptive management plan which is being developed concurrent with the HCP.

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Table 2.1. Optimal temperature ranges, optimal growth temperatures, maximum weekly average temperatures for growth, and maximum temperatures for survival of short exposures (24 hr) of different salmonids. Data are from Brungs and Jones (1977), Jobling (1994), and Pennell and Barton (1996).

| Species | Optimum range (°C) | Optimal growth temp (°C) | Maximum weekly average temp for growth (°C) | Maximum temp for survival of short exposure (°C) |
|---|---|---|---|---|
| Brook trout | 7.0-20.3 | 13.0-16.1 | 19.0 | 24.0 |
| Brown trout | 6.0-20.0 | 10.0-15.5 | 17.0 | 24.0 |
| Rainbow trout | 10.0-22.0 | 16.5-17.2 | 19.0 | 24.0 |
| Cutthroat trout | 12.0-15.0 | | | |
| Lake trout | 6.0-17.0 | | | |
| Coho salmon | 9.0-16.6 | 14.8 | 18.0 | 24.0 |
| Chinook salmon | 10.0-18.0 | 15.5 | | |
| Sockeye salmon | 10.0-15.0 | 15.0 | 18.0 | 22.0 |
| Pink salmon | 9.3-15.5 | 15.5 | | |
| Chum salmon | 13.0-14.1 | 13.0 | | |
| Arctic charr | 5.0-16.0 | 14.0 | | |

Table 2.2. Summary of temperature metrics (°C) compiled from 122 to 225 sites in 73 streams in Montana and Idaho.

| Statistic | Daily maximum temp[1] | Maximum daily average temp[2] | Maximum weekly maximum temp[3] | Maximum weekly average temp[4] | Monthly mean temp | |
|---|---|---|---|---|---|---|
| | | | | | July | August |
| Mean | 15.26 | 12.15 | 14.45 | 11.47 | 10.53 | 10.07 |
| Std dev. | 3.91 | 2.74 | 3.75 | 2.62 | 2.49 | 2.04 |
| Minimum | 8.29 | 6.89 | 7.96 | 6.77 | 6.56 | 6.43 |
| Maximum | 26.90 | 22.58 | 25.51 | 21.92 | 20.93 | 17.15 |
| Sample size | 225 | 218 | 124 | 124 | 122 | 123 |

[1] DMT is the maximum daily temperature measured on the hottest day of the year.
[2] MDAT is the maximum average daily temperature measured on the hottest day of the year.
[3] MWMT is the average of daily maximum temperatures over the warmest consecutive 7-day period.
[4] MWAT is the average of daily average temperatures over the warmest consecutive 7-day period.

Table 2.3. Pearson correlation coefficients between all possible combinations of temperature metrics compiled from 122 to 225 sites in 73 streams in Montana and Idaho. Temperature metrics DMT=daily maximum temperature, MDAT=maximum daily average temperature, MWMT=maximum weekly maximum temperature, MWAT=maximum weekly average temperature, JULY=mean July temperature, and AUG=mean August temperature.

| | DMT | MDAT | MWMT | MWAT | JULY | AUG |
|---|---|---|---|---|---|---|
| DMT | 1.00 | | | | | |
| MDAT | 0.93 | 1.00 | | | | |
| MWMT | 0.99 | 0.96 | 1.00 | | | |
| MWAT | 0.92 | 0.99 | 0.95 | 1.00 | | |
| JULY | 0.90 | 0.97 | 0.93 | 0.98 | 1.00 | |
| AUG | 0.91 | 0.98 | 0.94 | 0.98 | 0.96 | 1.00 |

Table 2.4. Summary of temperatures (°C) compiled from 68 to 70 sites in 33 streams supporting bull trout in Montana and Idaho.

| Statistic | Daily maximum temp[1] | Maximum daily average temp[2] | Maximum weekly maximum temp[3] | Maximum weekly average temp[4] | Monthly mean temp | |
|---|---|---|---|---|---|---|
| | | | | | July | August |
| Mean | 13.14 | 11.05 | 12.36 | 10.43 | 9.51 | 9.33 |
| Std dev | 2.71 | 2.10 | 2.52 | 1.99 | 1.78 | 1.56 |
| Minimum | 8.29 | 6.89 | 7.96 | 6.77 | 6.56 | 6.43 |
| Maximum | 19.72 | 15.82 | 18.73 | 15.23 | 14.09 | 13.19 |
| Sample size | 70 | 69 | 70 | 69 | 68 | 69 |

[1] DMT is the maximum daily temperature measured on the hottest day of the year.
[2] MDAT is the maximum daily average temperature measured on the hottest day of the year.
[3] MWMT is the average of daily maximum temperatures over the warmest consecutive 7-day period.
[4] MWAT is the average of daily average temperatures over the warmest consecutive 7-day period.

Table 2.5. Summary of water temperatures associated with salmonid life stages and behavior. Ranges given with more than one reference are a synthesis of those reported by each investigator.

| | Migration | Spawning (season) | Incubation | Juveniles (summer) | Optimum | Preferred | Lower lethal | Upper lethal |
|---|---|---|---|---|---|---|---|---|
| **Steelhead** | adults ≤ 21 [4], smolting stops > 13.7 - 17.6 [1] | 3.8 - 9.4 [2], 2 - 8 [8], spring | | 13 -19 [2], 7-14 [1] avoidance @ ≥ 22 [4] | 10 - 12.7 [2,11] | 7.2-14.4 [2] | | 23.9 [2,11] |
| **Rainbow trout** (subspecies not specified) | | 2.2 - 20, optimum 9.7 [1], spring | 5 - 15[1] | | 1 10 - 22 [2,11] | 13.8 [2] | 0 [2] | 29.4, avoidance @ > 21.6 [2] & @ 19-22 [3] |
| **Chinook salmon** (subspecies not identified) | 3.3-13.3 [2] (spring) | 5.5 - 14 [2,12] | 5 - 14.4[2] | 10 - 18 [11] | 10 - 18 [11] | 12 - 14 [12] | 0 [2] | 21.5 - 25 [2,11] |
| **Cutthroat trout** (subspecies not specified) | | 5.5 - 17.2 [2], optimum 10 [2,10], spring | 4.4-12.8[2] | | 12 - 15[11] | 9.4 - 12.7[2] | 0.5 [2] | 22.7 [2] |
| **Mountain Whitefish** | | 1.7-9.4, usually <6 [7], fall | | | | 12.2 - 16.6 [2], 9-11 [9] | | |

1. Barnhart 1991, in Stoltz & Schnell 1991.
2. Bell 1991.
3. Coutant 1977.
4. Lanz 1971, in Beschta 1987.
4. Mantelman 1960, in Beschta 1987.
6. Meehan & Bjornn 1991.
7. Northcote & Ennis 1994.
8. Orcutt et al. 1968.
9. Wydoski and Whitney 1979.
10. McIntyre and Reiman 1995.
11. Pennell and Barton 1996.
12. Bjornn and Reiser 1991.

Table 5.1. Summary of two-factor, mixed-model ANOVA with one repeated factor for the percentage of canopy cover over streams before and after timber harvest in 10 independent locations.

| Source | Degrees of Freedom | Mean Square Error | F-ratio | Probability level | Power ($\alpha$=0.05) |
|---|---|---|---|---|---|
| Survey Location | 9 | 2143.124 | 5.36 | 0.000 | 0.935 |
| Error | 100 | 399.918 | | | |
| Pre- & Post-Harvest | 1 | 1154.618 | 47.07 | 0.000 | 0.996 |
| Interaction | 9 | 132.186 | 5.39 | 0.000 | 0.936 |
| Error | 100 | 24.532 | | | |
| Total (adjusted) | 219 | | | | |

Table 5.2. Summary of paired t-tests for assessing differences in mean percent canopy cover before and after timber harvest in 10 independent survey locations.

| Survey location (Site #) | Mean canopy cover | | t-value | Probability level | Power ($\alpha$=0.05) |
|---|---|---|---|---|---|
| | Before | After | | | |
| 1 | 51.9 | 49.8 | 1.006 | 0.338 | 0.149 |
| 2 | 66.3 | 61.6 | 1.889 | 0.088 | 0.401 |
| 3 | 72.2 | 64.6 | 3.711 | 0.004 | 0.916 |
| 4 | 82.6 | 81.5 | 0.612 | 0.554 | 0.086 |
| 5 | 77.0 | 69.9 | 4.387 | 0.001 | 0.976 |
| 6 | 69.1 | 56.0 | 5.237 | 0.000 | 0.997 |
| 7 | 78.5 | 68.1 | 4.550 | 0.001 | 0.983 |
| 8 | 63.2 | 60.7 | 1.975 | 0.077 | 0.431 |
| 9 | 51.5 | 51.3 | 0.019 | 0.985 | 0.050 |
| 10 | 58.0 | 60.7 | -1.166 | 0.271 | 0.184 |
| **TOTAL** | **67.0** | **62.4** | | | |

Table 5.3. Summary of stream channel characteristics (n=10).

| | Mean | Range |
|---|---|---|
| Bankfull Channel Width (ft) | 11.2 | 7.7-24.1 |
| Wetted Channel Width (ft) | 6.5 | 0.7-16.9 |
| Average Water Depth (ft) | 0.5 | 0.1-1.4 |

Table 5.4. Riparian stand data for each survey site before and after harvesting.

| Location (Site #) | Stand Density (trees/ac) | | Mean Tree Height (ft) | | Mean Diameter (DBH) | |
|---|---|---|---|---|---|---|
| | Before | After | Before | After | Before | After |
| 1 | 259 | 181 | 77 | 78 | 12.5 | 12.2 |
| 2 | 105 | 93 | 78 | 78 | 18.1 | 17.6 |
| 3 | 126 | 77 | 75 | 73 | 14.3 | 13.7 |
| 4 | 81 | 70 | 86 | 81 | 17.3 | 15.9 |
| 5 | 290 | 201 | 71 | 69 | 11.4 | 11.2 |
| 6 | 145 | 69 | 97 | 93 | 15.7 | 15.0 |
| 7 | 223 | 142 | 74 | 72 | 15.2 | 14.4 |
| 8 | 93 | 73 | 89 | 90 | 18.1 | 18.0 |
| 9 | 122 | 91 | 81 | 79 | 17.7 | 16.1 |
| 10 | 207 | 172 | 69 | 70 | 12.7 | 12.5 |
| Mean | 165 | 117 | 80 | 78 | 15.3 | 14.7 |

Table 6.1. Mean, minimum, maximum and standard deviation for independent and dependent variables used in stream temperature regressions.

| | n | Mean | Standard Deviation | Minimum | Maximum |
|---|---|---|---|---|---|
| Elevation | 100 | 3534 | 675 | 1960 | 5670 |
| Canopy Cover | 100 | 53 | 21 | 9 | 100 |
| MWMT | 220 | 14.8 | 3.6 | 8.0 | 25.5 |
| MWAT | 217 | 12.9 | 3.2 | 6.8 | 22.1 |

Table 6.2. Regression statistics and coefficients for 13 regression models for Maximum Weekly Maximum Temperature (MWMT). N refers to sample size, r-square is the uncorrected multiple $r^2$, PSS is the average predicted sum of squares, and 'PI add' is the standard error added to the regression line for the prediction interval.

| | | | | Regression Coefficients | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Data set–Variables | N | $r^2$ | PSS | Y-Intercept | Elevation | Canopy Cover | Temperature | Drought Index | PI add |
| All data--Elevation, can cover | 220 | 49.2 | 6.87 | 27.34126 | -0.002072 | -0.09765 | | | 2.615 |
| 1994 Ony--Elevation, can cover | 38 | 76.6 | 4.66 | 29.30792 | -0.001394 | -0.13566 | | | 2.111 |
| 1995 Ony--Elevation, can cover | 82 | 41.5 | 6.19 | 24.00421 | -0.001482 | -0.08503 | | | 2.486 |
| 1996 Ony--Elevation, can cover | 38 | 56.1 | 7.46 | 30.73579 | -0.003113 | -0.08954 | | | 2.728 |
| 1997 Ony--Elevation, can cover | 45 | 48.4 | 6.70 | 29.18528 | -0.003015 | -0.07860 | | | 2.543 |
| | | | | | | | | | |
| All data-Elev, can cover, DI jul-aug | 220 | 57.7 | 5.78 | 27.14003 | -0.001870 | -0.09982 | | -0.89125 | 2.399 |
| All data-Elev, can cover, DI jun-aug | 220 | 57.0 | 5.87 | 27.10112 | -0.001868 | -0.10007 | | -1.35094 | 2.416 |
| All data-Elev, can cover, Temp jul-aug | 220 | 55.4 | 6.08 | 28.86229 | -0.002274 | -0.10205 | 0.24898 | | 2.462 |
| All data-Elev, can cover, Temp july | 220 | 55.4 | 6.09 | 28.33876 | -0.002184 | -0.09870 | 0.22588 | | 2.463 |
| | | | | | | | | | |
| All data-Elev, can cover, DI jun-aug, Temp jul | 220 | 58.8 | 5.66 | 27.76974 | -0.001987 | -0.10011 | 0.13794 | -1.01642 | 2.376 |
| All data-Elev, can cover, DI jul-aug, Temp jul | 220 | 58.8 | 5.68 | 27.70214 | -0.001976 | -0.09984 | 0.11642 | -0.67866 | 2.377 |
| All data-Elev, can cover, DI jul-aug, Temp jul-aug | 220 | 58.8 | 5.68 | 27.96791 | -0.002023 | -0.10155 | 0.12754 | -0.67556 | 2.378 |
| All data-Elev, can cover, DI jun-aug, Temp jul-aug | 220 | 58.8 | 5.68 | 28.07744 | -0.002041 | -0.10210 | 0.14982 | -1.00715 | 2.378 |

Table 6.3. Regression statistics and coefficients for 13 regression models for Maximum Weekly Average Temperature (MWAT). N refers to sample size, r-square is the uncorrected multiple $r^2$, PSS is the average predicted sum of squares, and 'PI add' is the standard error added to the regression line for the prediction interval.

| | | | | Regression Coefficients | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Data set–Variables | N | $r^2$ | PSS | Y-Intercept | Elevation | Canopy Cover | Temperature | Drought Index | PI add |
| All data–Elevation, can cover | 217 | 45.4 | 5.67 | 24.75364 | -0.002378 | -0.06600 | | | 2.374 |
| 1994 Only--Elevation, can cover | 38 | 69.3 | 4.34 | 26.18420 | -0.001739 | -0.09559 | | | 2.023 |
| 1995 Only--Elevation, can cover | 82 | 38.0 | 5.14 | 21.81585 | -0.001915 | -0.05420 | | | 2.259 |
| 1996 Only--Elevation, can cover | 35 | 49.2 | 6.47 | 27.50650 | -0.003204 | -0.06058 | | | 2.552 |
| 1997 Only–Elevation, can cover | 45 | 54.7 | 4.75 | 28.12300 | -0.003494 | -0.05868 | | | 2.169 |
| | | | | | | | | | |
| All data–Elev, can cover, DI jul-aug | 217 | 54.8 | 4.73 | 24.58006 | -0.002196 | -0.06792 | | -0.82252 | 2.170 |
| All data–Elev, can cover, DI jun-aug | 217 | 54.1 | 4.81 | 24.56863 | -0.002201 | -0.06826 | | -1.24177 | 2.188 |
| All data–Elev, can cover, Temp jul-aug | 217 | 52.2 | 5.00 | 26.09969 | -0.002550 | -0.06994 | 0.22804 | | 2.232 |
| All data–Elev, can cover, DI may-aug | 217 | 51.9 | 5.04 | 24.64702 | -0.002252 | -0.06856 | | -1.40674 | 2.241 |
| | | | | | | | | | |
| All data–Elev, can cover, DI jul-aug, Temp jul-aug | 217 | 56.0 | 4.64 | 25.29385 | -0.002325 | -0.06944 | 0.11398 | -0.62829 | 2.149 |
| All data–Elev, can cover, DI jun-aug, Temp jul-aug | 217 | 55.9 | 4.65 | 25.41515 | -0.002348 | -0.07003 | 0.13547 | -0.92735 | 2.151 |
| All data–Elev, can cover, DI jul-aug, Temp jul | 217 | 55.8 | 4.65 | 25.00421 | -0.002273 | -0.06785 | 0.09564 | -0.64680 | 2.152 |
| All data–Elev, can cover, DI jul-aug, Temp aug | 217 | 55.8 | 4.67 | 25.32527 | -0.002331 | -0.07044 | 0.09212 | -0.67782 | 2.157 |

Table 6.4. Average temperature deviations (°C) July and August from St. Maries, Idaho, Missoula, Montana and Kalispell, Montana, and drought severity indices for Montana and Idaho used in multiple regressions.

| Year | Temperature Deviation | | | ID Drought Index | | MT Drought Index | |
|---|---|---|---|---|---|---|---|
| | St. Maries ID (Jul, Aug) | Kalispell MT (Jul, Aug) | Missoula MT (Jul, Aug) | Jun-Aug | Jul-Aug | Jun-Aug | Jul-Aug |
| 1991 | 0.8, 3.7 | 0.0, 5.1 | 2.5, 5.5 | 0.60 | 0.46 | 1.70 | 1.57 |
| 1992 | -6.5, -1.9 | -5.2, 1.6 | -4.6, 1.7 | -3.82 | -3.62 | -1.75 | -1.33 |
| 1993 | -14.2, -8.0 | -13.0, -4.2 | -11.9, -3.6 | 2.00 | 2.67 | 2.26 | 3.52 |
| 1994 | -2.2, 1.4 | -4.5, 5.3 | 5.6, 7.8 | -3.60 | -3.65 | -2.27 | -2.49 |
| 1995 | -4.6, -7.3 | -3.7, -4.1 | -0.6, 0.1 | 1.85 | 2.17 | 1.43 | 1.97 |
| 1996 | -0.5, -2.6 | 1.7, 1.2 | 4.2, 1.4 | 0.63 | 0.05 | 3.05 | 2.55 |
| 1997 | -5.8, -1.8 | -3.4, 1.8 | -2.9, 1.2 | 1.68 | 2.01 | 4.29 | 4.31 |

Table 6.5. Percentiles for modified Palmer drought indices (DI) for Montana (region 1) and Idaho (region 4) for the period 1895-1997. Indices are monthly averages for the period June-August. Reoccurrence interval (yr) is calculated as: 100/percentile. Percentiles of lower and upper confidence intervals are shown for the DI.

| Percentile | Montana | | | Idaho | | |
|---|---|---|---|---|---|---|
| | DI Value | Lower c.i. | Upper c.i. | DI Value | Lower c.i. | Upper c.i. |
| 50 | 0.68 | 0.13 | 1.18 | 0.21 | -0.28 | 0.63 |
| 45 | 0.4 | -0.47 | 1.01 | -0.09 | -0.85 | 0.44 |
| 40 | 0.12 | -0.86 | 0.68 | -0.3 | -1.07 | 0.21 |
| 35 | -0.48 | -1.24 | 0.34 | -0.87 | -1.37 | -0.11 |
| 30 | -0.94 | -2.06 | 0.11 | -1.17 | -2.09 | -0.59 |
| 25 | -1.49 | -2.44 | -0.59 | -1.46 | -2.25 | -0.93 |
| 20 | -2.09 | -2.69 | -1.15 | -2.11 | -2.5 | -1.23 |
| 15 | -2.6 | -4.19 | -1.75 | -2.33 | -3.34 | -1.79 |
| 10 | -3.07 | -5.03 | -2.44 | -2.66 | -3.6 | -2.25 |
| 5 | -4.95 | -6.06 | -3.09 | -3.58 | -4.28 | -2.7 |
| 1 | -6.03 | | | -4.26 | | |

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Figure 2.3. Relationships between maximum weekly average temperatures (MWAT) and maximum daily and average daily temperatures compiled from 122 to 225 sites in 73 streams in Montana and Idaho.

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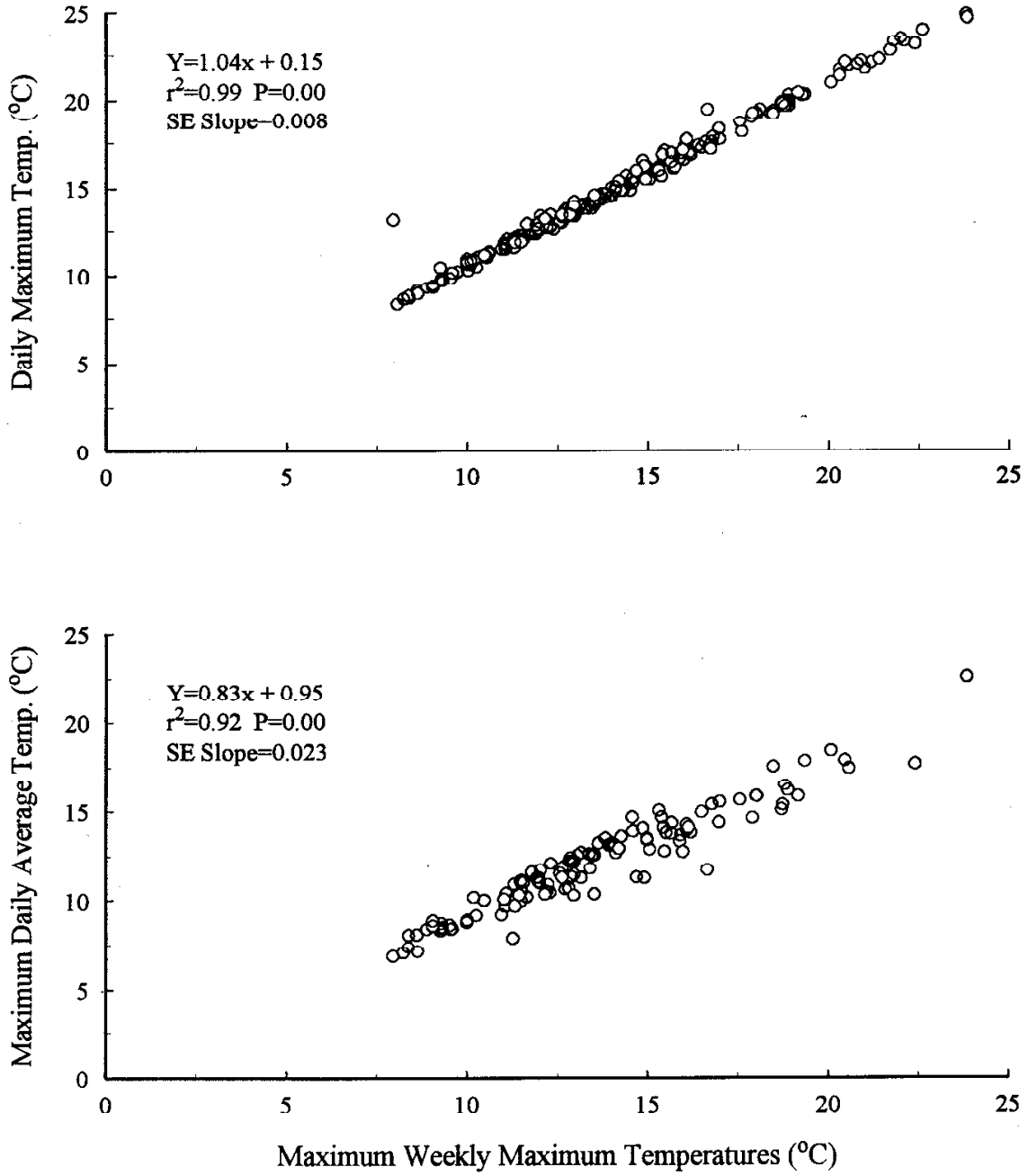
Figure 6.3. Predicted maximum weekly maximum temperatures (MWMT) as a function of canopy cover and elevation ($r^2$=0.58, DI=0).

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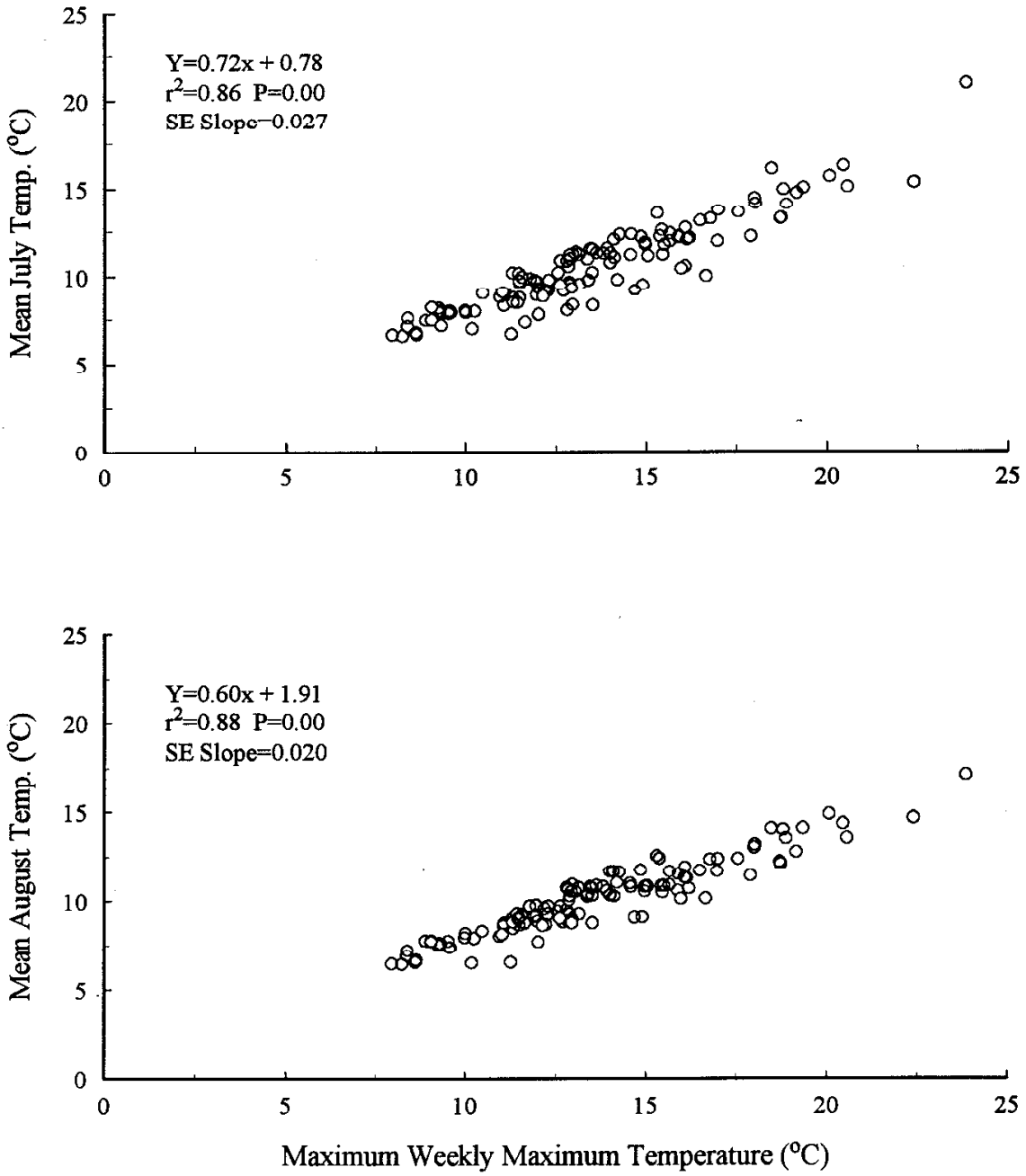



Figure 2.2. Relationships between weekly maximum temperatures (MWMT) and average July and August temperatures compiled from 122 to 225 sites in 73 streams in Montana and Idaho. Simple linear regression results are shown.

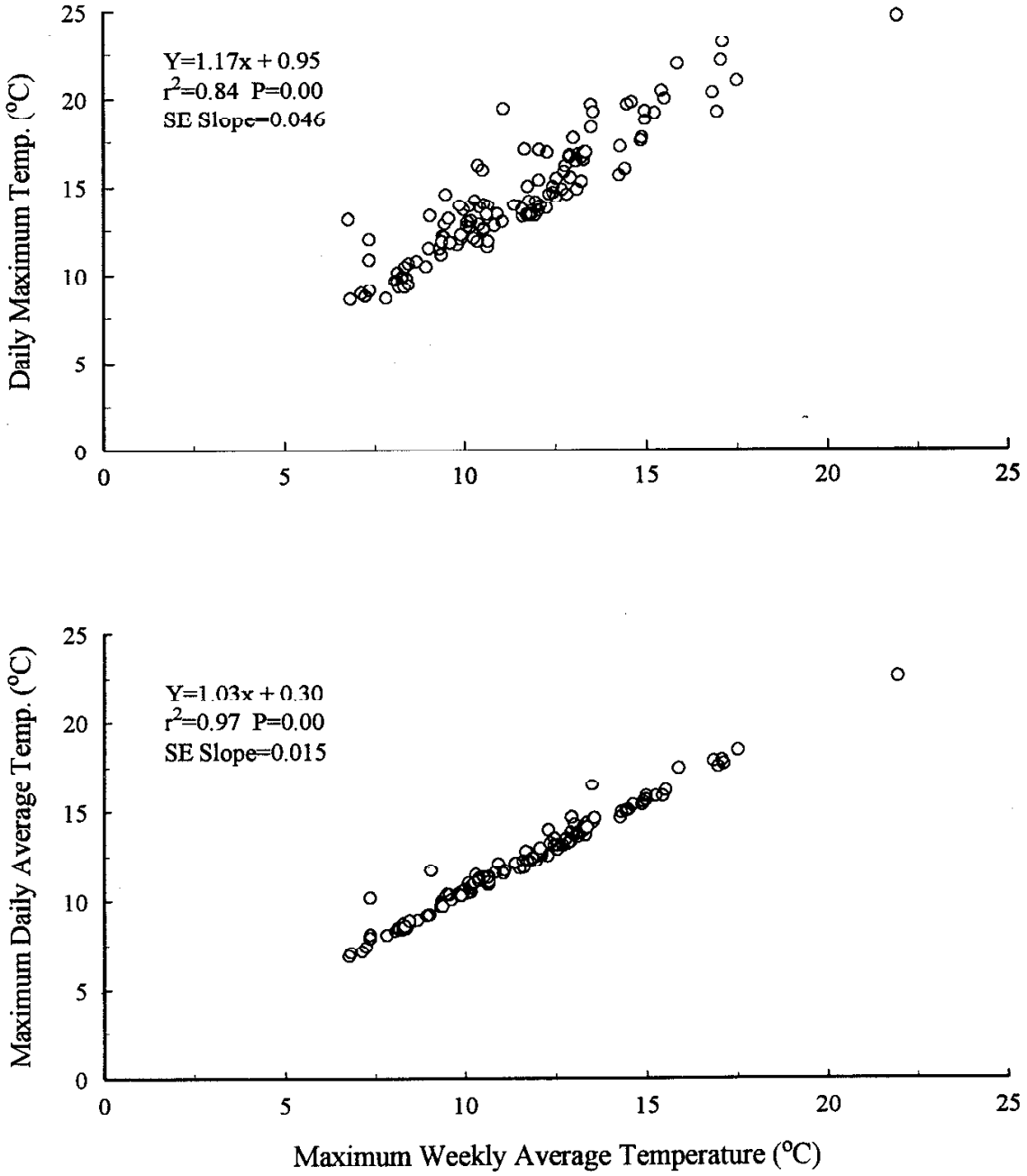
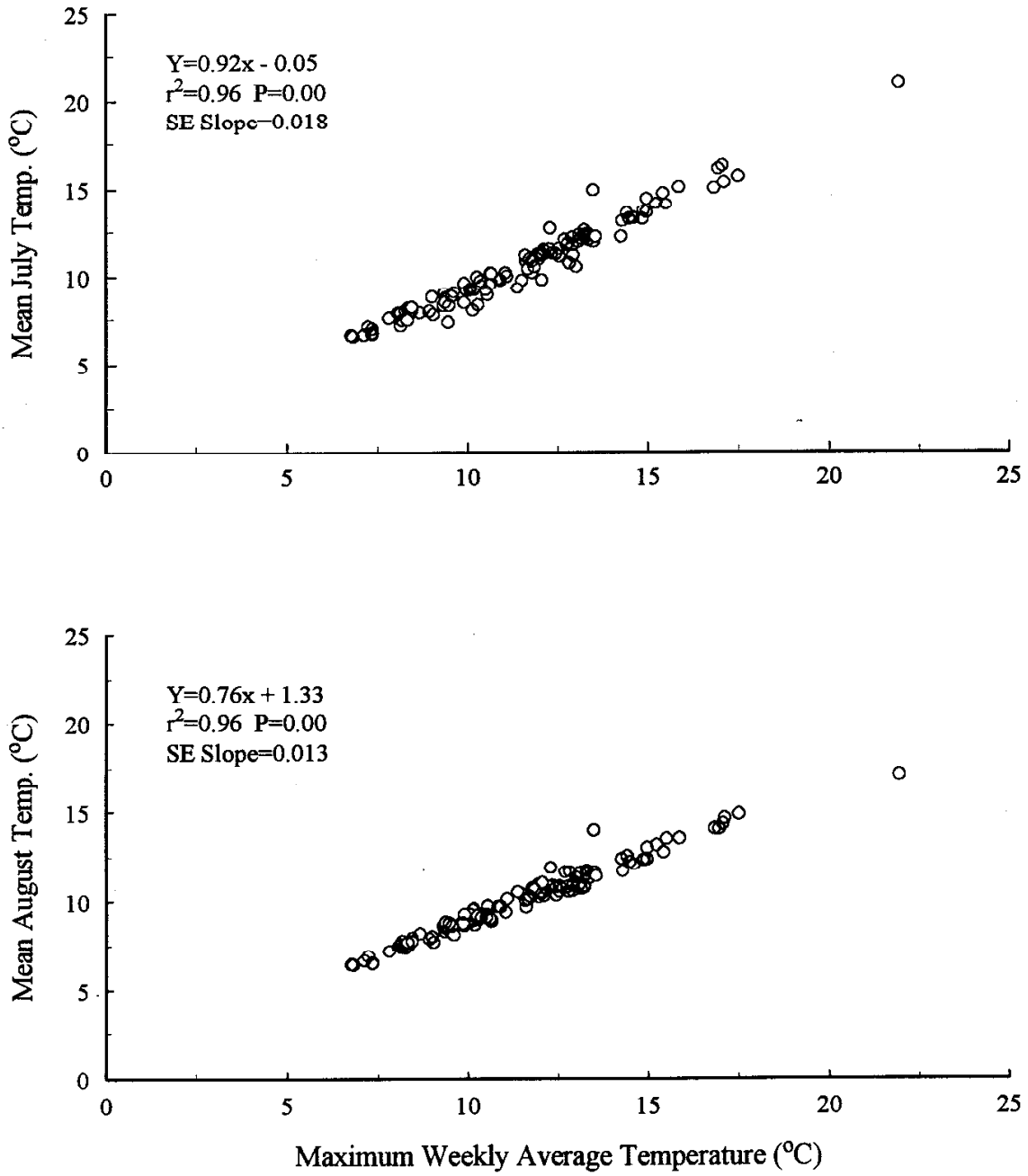



Figure 2.3. Relationships between maximum weekly average temperatures (MWAT) and maximum daily and average daily temperatures compiled from 122 to 225 sites in 73 streams in Montana and Idaho. Simple linear regression results are shown.

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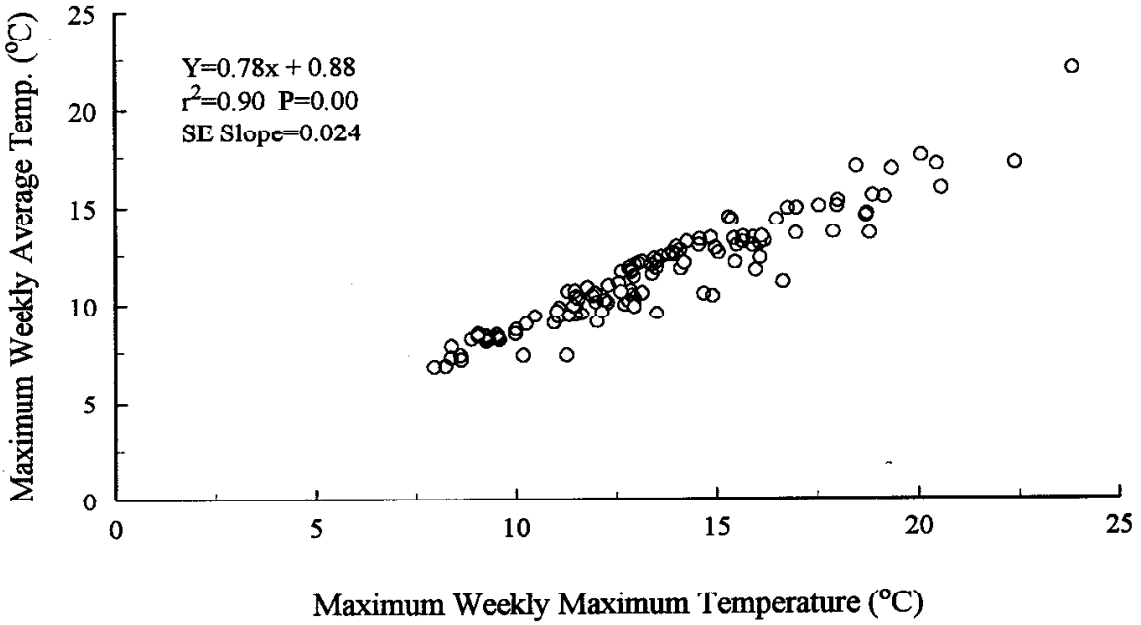


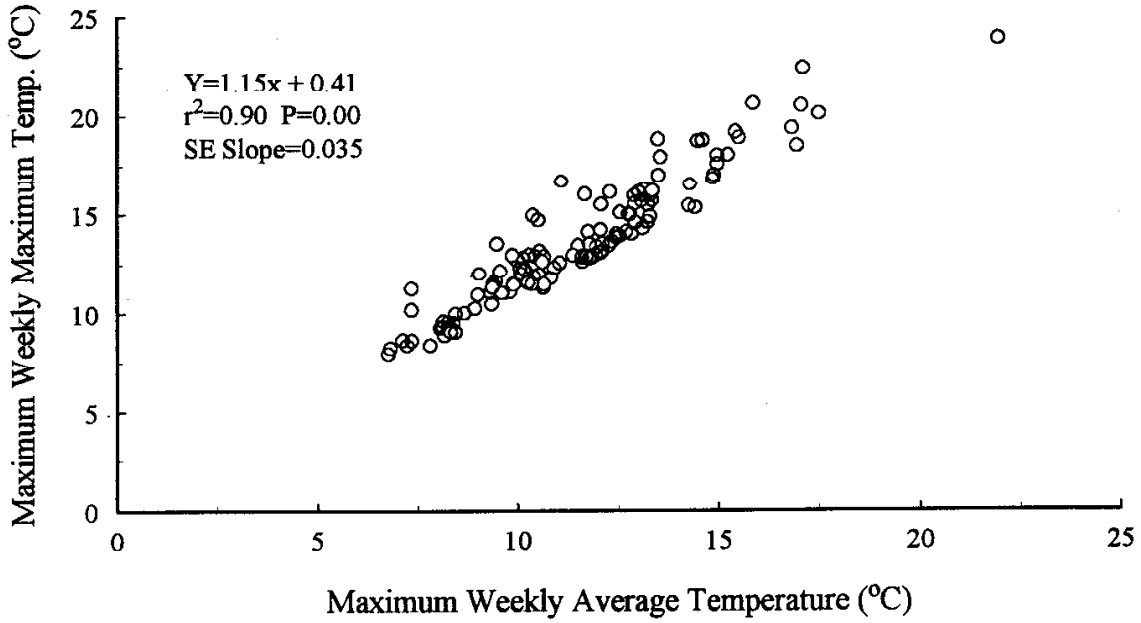


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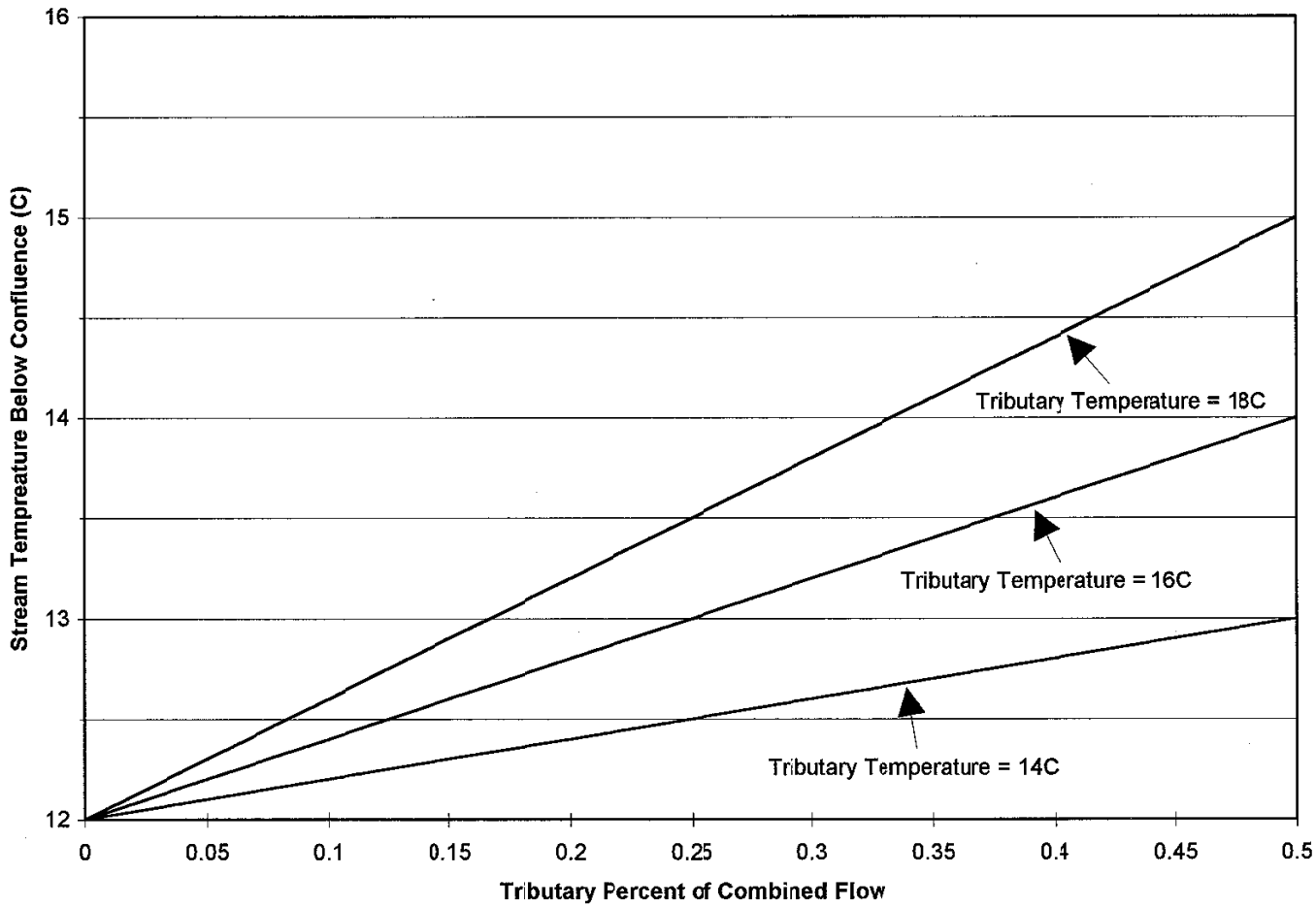
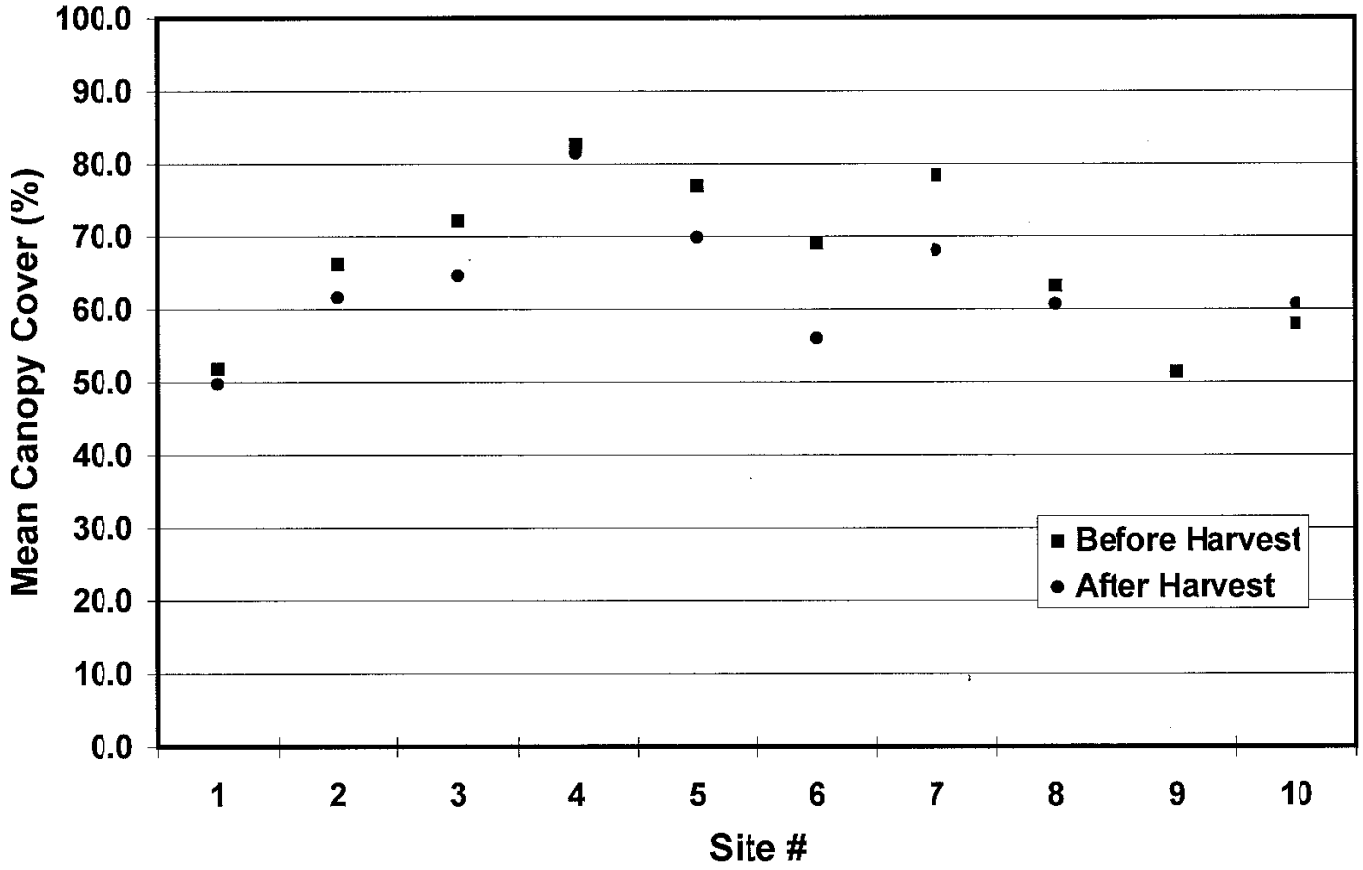


Figure 5.1 Mean canopy cover before and after harvest for each survey site. Sites 3, 5, 6, and 7 had a statistically-significant decrease in canopy cover ($p<0.05$)



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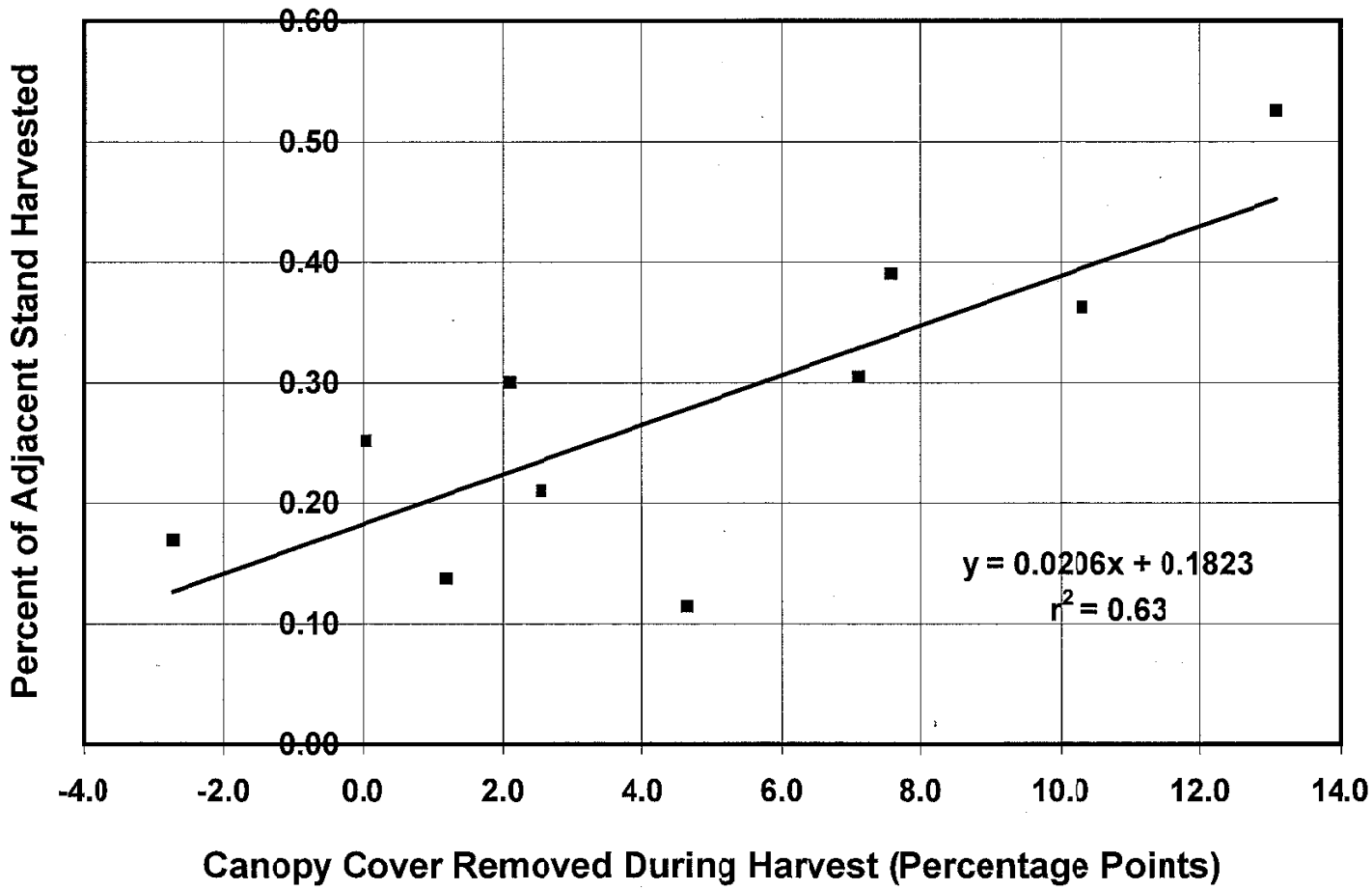

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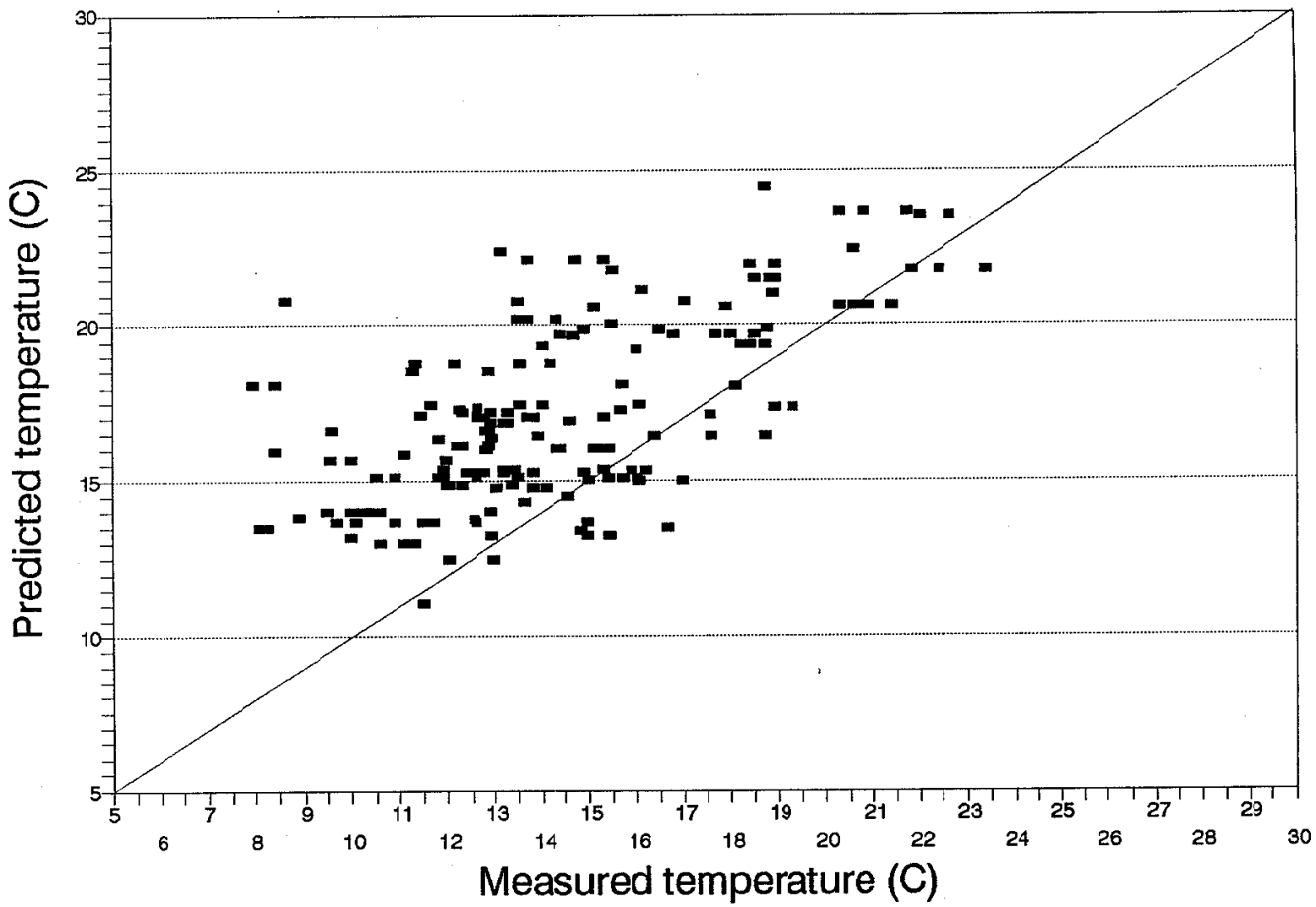

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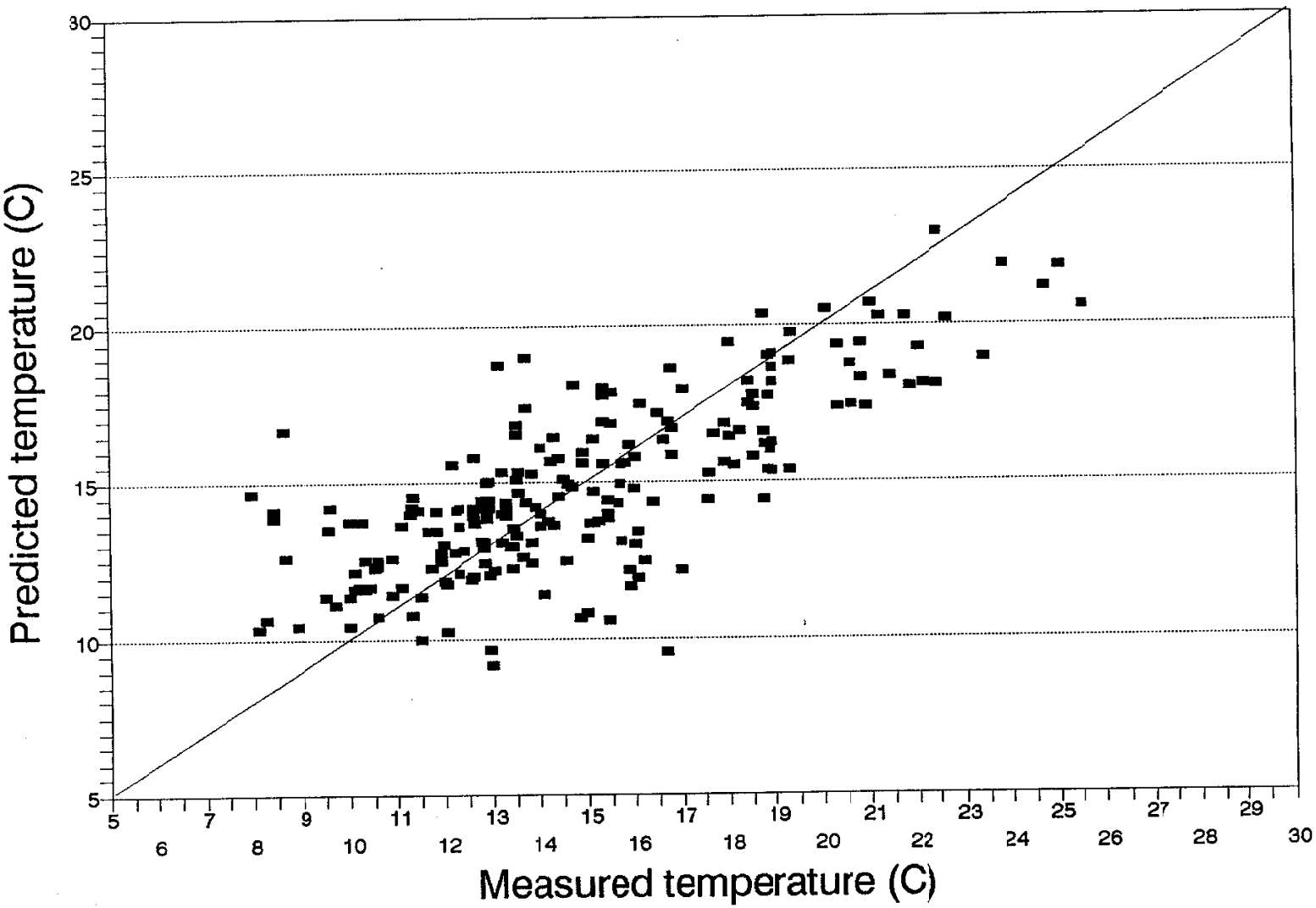
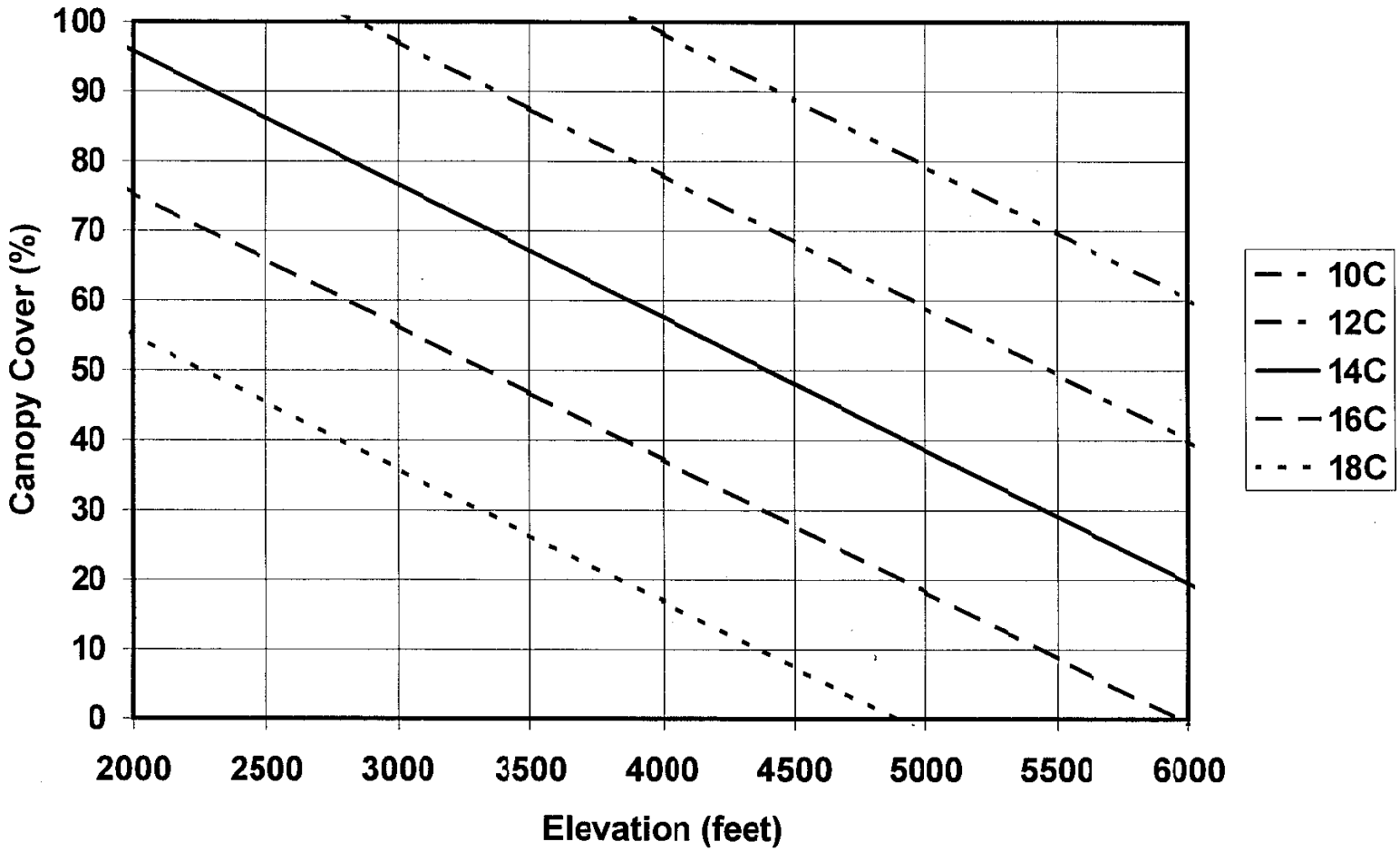
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