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STREAM TEMPERATURE INDICES, THRESHOLDS, AND STANDARDS USED TO PROTECT COHO SALMON HABITAT: A REVIEW

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March 2002

1 Introduction

Most fish maintain body temperatures that closely match their environment (Moyle 1993). As a result, temperature has a strong influence on almost every life history stage of Pacific salmon and steelhead (Berman 1998), including metabolism, growth and development, timing of life history events such as adult migration and emergence from the redd, and susceptibility to disease (Groot et al. 1995). There are several good recent reviews available on the effects of temperature on salmonids, including Groot et al. (1995), Sullivan et al. (2000), and McCullough (1999).

Exposure to high temperatures can have a variety of adverse effects on the physiology and physical performance of salmonids. These effects may vary depending on a fish's prior thermal history. Figure 1 shows some of the types of assays that have been used to evaluate the adverse effects of high temperatures. They fall into two general categories: effects on aspects of physical performance and effects on physiological tolerance (including growth).

There have been several laboratory studies to determine the high temperatures that lead to mortality of juvenile salmonids. However, there is some question about whether the temperature thresholds and effects identified in a carefully controlled laboratory setting are applicable to natural environments, which are much more dynamic and variable. Daily and seasonal variability in stream water temperatures also makes it difficult to define water temperature standards that are protective of salmonids. It is even more difficult to identify temperature standards that protect against sublethal effects upon salmonids, such as reduced growth.

The goal of this paper is to begin developing a strategy to address the cumulative effects of land management practices on stream temperature and, by extension, anadromous salmonid populations, particularly coho salmon. This report reviews the most commonly employed indices of temperature impairment and the methods used to set thresholds for these indices. This report then discusses potential approaches for developing temperature indices and thresholds that could be used to define regulatory temperature standards appropriate for northern California, and how to determine where in a watershed these standards should be applied to protect anadromous salmonid populations in lands actively managed for timber production.

This paper focuses on the juvenile rearing period for coho salmon to evaluate temperature indices and thresholds. This analysis focuses on coho salmon because they are generally more sensitive to warm stream temperatures than steelhead trout (Bjornn and Resier 1991). We focus on the juvenile rearing period for coho salmon because summer temperatures are generally warmer in northern California than in the more northerly range of Pacific salmonid stocks. Juvenile coho salmon thus serves as an indicator species and life history stage since they are most sensitive to warm temperatures of Pacific salmonids

2 Stream Temperature Indices

This section describes indices typically used for evaluating stream temperature, and will evaluate the strengths and weaknesses of each. For the purposes of this report we define an **index** as a means of summarizing temperature data that are related to a biological objective. We define **thresholds** as the value of an index that temperature must remain below to avoid adverse impacts. **Standards** then, are defined as a combination of an index and threshold(s), which can be applied by a regulating body. Established temperature standards, including those included in state water quality regulations in California, Oregon, and Washington for the protection of rearing salmonids, will be discussed with regard to each index. Our focus is on evaluating the biological relevance of temperature indices and thresholds.

2.1 Annual maximum temperature

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The annual maximum index is the maximum daily temperature that occurs each year. The objective of the annual maximum temperature index is to protect against short-term temperature increases that can result in direct mortality. Brett (1952) conducted laboratory experiments to determine upper incipient lethal temperatures for a variety of salmonids, which continue to be relied upon in the literature for providing the basis for thresholds using the annual maximum index. In a controlled environment it is possible to acclimate fish to varying temperatures, determine lethal temperatures (usually the temperature at which 50% of the population dies for a given acclimation temperature), and determine optimal temperatures for growth with varying rations. Brett (1952) studied the temperature tolerances of Washington state hatchery stock salmonid fry in a series of abrupt transfer experiments. Brett (1952) determined that the upper incipient lethal temperature UILT (the temperature at which 50% of fry would be expected to survive indefinitely) varied with acclimation temperature as follows (result shown here are for coho salmon fry):

Acclimation temperature	UILT
5°C	22.9°C
10°C	23.7°C
15°C	24.3°C
20°C	25.0°C
23°C	25.0°C

The UILT for acclimation temperatures of 20ºC and 23°C resulted in the highest temperature tolerance (25.0°C), and was a reasonable estimate of the "ultimate upper incipient lethal temperature" (UUILT) which is the maximum UILT attainable for any acclimation temperature. The 25°C has been widely cited in the literature as the UILT for juvenile coho salmon (e.g., ODEQ 1995, Brungs and Jones 1977). Bjornn and Reiser (1991) and Spence et al. (1996) both refer to a UUILT of 26°C and cite Brett (1952), but it appears that this may be an error.

Although setting maximum temperature standards is crucial to protect against potential lethal temperature effects, the results of laboratory-based studies may not apply to site-specific situations in the natural environment. Upper lethal temperatures in streams, unlike laboratory conditions, can be influenced by local genetic or physiological adaptations, food availability, varying or unknown acclimation temperatures, or access to cool water refugia.

Thresholds

Maximum instantaneous temperature thresholds are used to address temperatures that may result in direct mortality. In the zone of resistance (i.e., for temperatures above UILT), mortality is a function of exposure time; therefore, thresholds for maximum temperatures should address the duration of exposure.

Brungs and Jones (1977) begin with a formula for estimating LT50 (the temperature producing 50% mortality at a given exposure time) that is a function of the amount of time exposed within the zone of resistance.

$$LT50 = (\log_{10} \text{exposure time} - a)/b$$

where a and b are regression coefficients determined from laboratory experiments. Citing a number of studies, they assert that a 2°C reduction from an incipient lethal temperature generally yields no observed mortality and is a "safe" temperature for many sublethal effects. They apply this safety factor in their final recommendation:

$$\text{temperature} < (\log_{10} \text{exposure time} - a)/b - 2°C$$

whenever temperatures exceed UILT - 2°C.

The EPA has used the above formula combined with a one-hour exposure time to set maximum (instantaneous) annual temperatures, using the laboratory work of Brett (1952) for parameters a and b (at UILT of 25°C). Current water quality regulations for the state of Washington (WDOE 2000) included an annual 1-day maximum temperature standard of 16°C (Table 1) to protect against lethal effects. No basis for the State of Washington standards are reported (WDOE 2000), though they are markedly lower than the EPA's. Since the Washington standards are based on a 1-day temperature duration, it is possible that an UILT could be exceeded for one hour and stream temperatures would not be considered in violation of state standards.

Sullivan et al. (2000) used the same basic equation as Brungs and Jones, relating LT50 to exposure time to justify their recommendations for maximum permissible instantaneous temperatures, but they do not include the 2°C protection, presumably because they regard the sublethal effects as adequately addressed by their primary MWAT threshold (discussed below). They modified the formula to predict LT10 (the temperature producing 10% mortality at a given exposure time) instead of LT50, citing laboratory studies that showed the LT10 to be at least 98% of LT50 for the same exposure time. Applying this formula to their study locations whenever temperatures exceeded 24°C, they concluded that a maximum annual temperature of 26°C provided adequate protection against direct mortality. However, this conclusion depended on durations of high temperatures typical in their study area, and they recommended conducting site-specific investigations of temperature patterns when temperatures exceed 24°C.

While an annual maximum standard based on lethal effects may be useful for preventing acute temperature effects (mortality), an upper threshold alone will not ensure the long-term health of a fish population (ODEQ 1995, Sullivan et al. 2000). Therefore, most temperature standards also include a lower threshold that is intended to prevent temperature-related reductions in growth, or other vital functions. The state of Washington has dropped the annual maximum index from its proposed water quality standards, and has instead included indices that are believed more likely to ensure long-term population viability (WDOE 2000).

Table 1. Temperature standards for juvenile coho salmon reported in the literature.

Source	Basis for Threshold	Metric Used[1]	Acute	Sublethal	Comments
Brungs and Jones (1977)	Laboratory study	MWAT		18°C	based on laboratory data from Brett (1952) and Great Lakes Research laboratory (1973). Used by the EPA to establish standards
		Annual 1-day maximum	24°C		
NMFS and USFWS (1997)	Laboratory study	MWAT		16.8°C	using procedure from Brungs and Jones (1977), with optimal temperature data from Reiser and Bjornn (1979)
Eaton et al. (1995)	Field study	MWAT		23.4°C	upper thermal tolerance, from 193 streams throughout the United States
Welsh et al. (2001)	Field study	MWAT		16.7°C	upper thermal tolerance in 21 tributaries to the Mattole River
		MWMT		18.0°C	
Hines and Ambrose (unpublished)	Field study	MWAT		16.8°C	upper thermal tolerance in northern California streams
		MWMT		18.3°C	
Sullivan et al. (2000)	Risk-based	MWAT		14.8°C	growth analysis threshold for maintaining growth within 10% of optimum
		MWMT		16.5°C	
		Instantaneous maximum (1-hour)	26°C		
Sullivan et al. (2000)	Risk-based	MWAT		19°C	growth analysis threshold for maintaining growth within 20% of optimum
		MWMT		20.5°C	
ODEQ (1995)	Literature review based	MWMT		17.8°C	
WDOE (2000)	Not stated	Annual 1-day maximum	16°C		existing standard
WDOE (2000)	Literature review based	MWMT		17.0°C	proposed standard
EPA pers. comm. 1997	not stated	MWAT		17.1°C	as cited in CDF 1997

1 MWAT= maximum weekly mean temperature
MWMT= maximum weekly maximum temperature

2.2 Maximum weekly average temperature (MWAT) and maximum weekly mean temperature (MWMT)

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The use of maximum weekly average temperature (MWAT) was first proposed by the National Academy of Sciences (NAS and NAE) in 1972 (NAS and NAE 1973) as a long-term standard for preventing chronic sublethal effects for a variety of fish species. In its simplest interpretation, MWAT is the mathematical mean of multiple, equally spaced daily temperatures measured over a 7-day period (Brungs and Jones 1977). The MWAT however, is not calculated consistently by different researchers and agencies. For example, NMFS and USFWS (1997) and the EPA calculate the MWAT as the highest 7-day period calculated as an average of daily mean temperature. In contrast, the Oregon Department of Environmental Quality (ODEQ) and Sullivan et al. (2000) report MWAT as the highest 7-day period as calculated as an average of daily maximum temperatures, also referred to in the literature as the maximum weekly maximum temperature (MWMT) (Welsh et al. 2001). MWAT (or MWMT) is currently a convenient way to compare the results of researchers, and is the most commonly used index for establishing temperature standards for salmonids.

Thresholds

The objective of the MWAT index is to provide an upper temperature standard that is protective of juvenile salmonids during the summer rearing period. Brungs and Jones (1977) prepared a report for the EPA that used the procedure developed by the NAS and NAE (NAS and NAE 1973) to calculate MWAT thresholds for use as a temperature standard. Brungs and Jones (1977) calculated this MWAT threshold with the intention of protecting juvenile salmonids from adverse reductions in growth. Growth of salmonids is highly related to temperature and ration size. Optimal temperatures and UUILTs have been determined in laboratory studies by controlling temperature, acclimation temperature, and ration size. MWAT in this context was calculated as:

$$\mathrm{MWAT} = \mathrm{OT} + \frac{\mathrm{UUILT} - \mathrm{OT}}{3}$$

where:
OT= optimal temperature for a particular life stage or function
UUILT = the upper incipient lethal temperature (see section 2.1)

While the above equation has been used repeatedly (e.g., ODEQ 1995, NMFS and USFWS 1997), very little data exist to support its use for establishing standards to protect aquatic species. When this equation was developed by NAS and NAE (1973) they believed they had developed a useful method for estimating temperatures that may be limiting survival or growth of salmonids. The basis for their calculation was the observation that the maximum temperature at which several species are observed in nature lies near the average of their optimum temperature and the temperature at which growth does not occur with maximum rations (zero net growth). Further, they observed that for several species (i.e., channel catfish, white sucker) optimum growth rate at this same maximum temperature would generally be reduced to no lower than 80% of the maximum. This range, NAS and NAE (1973) concluded, appeared acceptable, though they noted it was not based on quantitative studies, or upon a specific impact to populations. When the NAS and NAE (1973) report was prepared there was little existing data on zero net growth, so for practical considerations they attempted to develop a similar relationship using the UILT, which they did in the MWAT calculation. They found that the MWAT equation (shown above) yielded values very close to the average of optimum temperature and zero net growth, and thus concluded they developed a practical method for obtaining thresholds. The reliance on the MWAT index that has occurred since the early 1970s has led to the implicit assumption that the MWAT index is biologically meaningful; however, there is little evidence to support this assumption.

The biological basis for the MWAT continues to be the assumption that it will ensure that growth rates for individuals in a given population will not be reduced by more than 20% of optimum growth. The 20% threshold continues to be based on no specific level of impairment, and thus there is no reason to assume that it is realistic, or meaningful.

2.2.1 Laboratory-based MWAT thresholds

The EPA established MWAT temperature standards for salmonids based on Brungs and Jones (1977). The specific data that Brungs and Jones used to establish their MWAT standard is not clear from their report, but it appears that they selected an optimum temperature between 14.5° and 15°C citing Great Lakes Research Laboratory (1973), and Averett (1968), and an upper lethal threshold between 24° and 25°C based on Brett (1952). The Great Lakes Research Laboratory report (1973) did not contain the methods used to derive an optimum temperature for coho salmon. It appears they had conducted experiments on coho salmon, but the details of these experiments (e.g., age of fish, sample size) were not reported. Other researchers have reported optimum growth rates for coho salmon on full rations as occurring at 15°C (e.g., Everson 1973, as cited in McCullough et al. 2001). Using the temperature data from these laboratory experiments, Brungs and Jones (1977) established an MWAT standard for sublethal effects on coho salmon of 18°C, but they did not conduct a specific analysis to determine the biological relevance of this standard.

The National Marine Fisheries Service (NMFS and USFWS 1997) also used the procedure developed by Brungs and Jones (1977) to determine an MWAT standard for their properly functioning conditions (PFC) matrix prepared for the Pacific Lumber Company Habitat Conservation Plan. NMFS and USFWS (1997) used a lower optimal temperature for juvenile coho salmon of 13.2°C (based on Reiser and Bjornn 1979) and thus calculated a lower MWAT of 16.8°C. Reiser and Bjornn (1979) did not suggest that 13.2°C was an optimum temperature, however, but rather reported an optimum temperature from Brett et al. (1958) of 20°C, and a range of preferred temperatures from Bell (1973) from 11.8 to 14.6°C. The selection of 13.2°C as an optimum temperature by NMFS and USFWS (1997) was not explained, and seems too conservative given that other researchers typically report optimum temperatures for juvenile coho growth as high as 14°C (Brett 1952). Actual optimum temperatures for juvenile salmonids in natural streams will depend on site-specific factors such as food availability.

Laboratory experiments have the advantage of allowing temperatures and rations to be controlled, often producing significant relationships between temperature, rations, and growth rates. Wild fish, however, exhibit complicated responses to water temperature, and respond to site-specific variables that are absent from controlled experiments. It is evident that even when the same procedure (i.e., Brungs and Jones 1977) is used to establish MWAT thresholds, the choice of optimum temperature has led to a variety of different standards. Professional judgment appears to be the dominant method used for selecting the optimum temperature data used for calculating standards, leading to a variety of interpretations and thresholds, none of which will necessarily protect fish in a predictable manner. Further, the MWAT approach used by NMFS and USFWS (1997) and Brungs and Jones (1977) have no component to allow for regional variation, and do not account for the role of food availability (Ligon et al. 1999).

2.2.2 Field-based MWAT thresholds

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MWAT temperature thresholds may also be based on data examining the temperature tolerances of fish in natural systems. The approach that is generally used is to determine the cutoff in MWAT temperature based on presence or absence for a particular fish species. Tolerance of high temperatures in a laboratory setting can only serve as an approximation of the thermal tolerances of fish under natural conditions, whereas a field-based approach can account for possible food limitations, or interactions with other species. This index also has the advantage of having a clear and simple biologically meaningful objective (fish presence). Standards developed in this manner, however, do not address sub-lethal effects, and may not ensure adequate summer growth.

Thresholds

Eaton et al. (1995) examined data from 193 streams throughout the United States with and without coho salmon to estimate the maximum temperature tolerance of juvenile coho salmon, expressed as an MWAT. A bootstrap method was used to estimate a standard error. Using this method, Eaton et al. (1995) reported an upper thermal limit for coho salmon as an MWAT of 23.4°C, with a standard error of 0.23°C.

Welsh et al. (2001) studied stream temperatures and presence/absence data for juvenile coho salmon during late summer in 21 tributaries to the Mattole River of northwestern California. The warmest tributaries that supported juvenile coho had an MWMT of 18.0°C or less, or an MWAT of 16.7°C or less. Juvenile coho salmon were found in all streams sampled with an MWMT of 16.3°C or MWAT of 14.6°C or less. This data contrasts sharply with the results of Eaton et al. (1995), which reported a much higher thermal tolerance for juvenile coho. Though Eaton et al. (1995) included almost 200 streams in their data set, Welsh et al. (2001) note that most of the study reaches were located within large rivers, where temperatures tend to be higher. This suggests that much of Eaton et al.'s (1995) data would be based on portions of the river that may be serving primarily as migration corridors for coho salmon, rather than rearing habitat, and this may have resulted in a higher upper thermal limit. Differences in results of these two studies, which were obtained using similar approaches, indicate the field-based approach is sensitive to location in stream network, or that temperature tolerances may vary geographically.

Welsh et al. (2001) caution that using their MWAT threshold as a standard should not occur without considering historical thermal regimes that existed in streams in the absence of management activities. It is likely that natural variation in vegetation communities and natural disturbance regimes in some watersheds have historically limited the use of some streams by coho salmon.

Hines and Ambrose (unpublished report) conducted an analysis similar to Welsh et al. (2001) for 32 sites in several streams located in the range of the Central California Coast Coho Salmon ESU. They concluded that streams with an MWMT of 18.3°C or higher and an MWAT of 16.8°C or higher were highly unlikely to support juvenile coho. Their results are remarkably similar to those of Welsh et al. (2001). In addition, Hines and Ambrose found that the number of days an MWMT of 17.6°C was exceeded was the most significant MWMT threshold examined in a model used to predict coho presence or absence. An additional advantage to both the Hines and Ambrose (unpublished) and Welsh et al. (2001) reports is that they are based on data collected in northern California, and are therefore more likely to have useful applications for this region.

A potential source of error for all field-based approaches is the study design. It is crucial that the only differences between study sites with, and those without presence, are related to temperature. If other habitat characteristics (e.g., large woody debris volume, drainage area, channel morphology, fine sediment, food availability, instream flows) are even partially limiting the species distribution, then the study is potentially flawed. Welsh et al. (2001) attempted to sample all streams accessible to coho, and did not control for other habitat characteristics. Hines and Ambrose (unpublished report) attempted to address habitat variables, but their habitat data were limited. In addition, the presence of spawning pairs and density of fry before the summer were not reported in any of the studies. It is possible that some of the sites with reported absence never had spawners due to limiting factors other than temperature. Another potential source of error is the possibility that fish are utilizing thermal refugia, and are therefore actually inhabiting areas with lower temperatures than those recorded for that stream. Thermographs placed in riffles, for example, may fail to capture the temperatures occurring in a shaded pool, or under an undercut bank. This is less likely to be a source of error in systems where thermal mixing occurs.

The field-based approach as described here ignores the condition of fish. It is likely that fish could be present at low densities in a stream, which would count as "presence," but their condition (i.e., general health and growth ability) could be low enough to prevent subsequent survival. Use of density measurements, growth rates, or condition factor may be more appropriate than presence to ensure that appropriate protective thresholds are selected. Li et al. (1993, as cited in McCullough et al. 2001) for example, reported that even though fish were still present, a significant decline in steelhead biomass occurred when temperatures in the John Day River increased from 16°C to 28°C.

2.2.3 Review-based approach

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MWAT standards have also been based on reviews of thresholds reported in the scientific literature. Sullivan et al. (2000) states that Bell (1973) conducted one of the first reviews of temperature standards for salmonids for the purpose of making threshold recommendations. A descriptive method for making the recommendations, or citations to the specific thresholds used, were lacking in Bell (1973). Therefore Sullivan et al. (2000) notes that it is not possible to review the merit of the recommendations made by Bell (1973), or by the updated Bell (1986). Despite the obvious drawbacks to reporting thresholds recommended by Bell (1973), Reiser and Bjornn (1979), Bjornn and Reiser (1991), Spence et al. (1996), and the U. S. Fish and Wildlife Service (Laufle et al. 1986, Pauley et al. 1986, as cited in Sullivan et al. 2000) all present the recommendations of Bell (1973) or Bell (1986) as temperature requirements for salmonids. Bjornn and Reiser (1979, 1991) have been cited in subsequent temperature threshold recommendations as forming at least part of the basis for proposed standards (Armour 1991, ODEQ 1995, NMFS and USFWS 1997). This trend of ignoring primary literature has led to data being extrapolated beyond their appropriate application, and has promoted continuing transmission of errors.

Thresholds

In 1995, the Oregon Department of Environmental Quality (ODEQ 1995) reviewed available literature for the purpose of developing MWMT standards for the state of Oregon. Sullivan et al. (2000) notes that the ODEQ relied entirely on literature reviews, such as Bjornn and Reiser (1991). An MWMT threshold of 17.8°C was selected, although no link was made from the literature review to the selected threshold, and no data were offered to support the standard.

The Washington State Department of Ecology (WDOE) currently has a 1-day maximum temperature standard of 16°C that applies to what they define as "extraordinary salmon spawning" streams. The WDOE, under pressure from private industry and environmentalists, has conducted a review of the literature to update their standards (WDOE 2000). They concluded that to fully protect juvenile coho salmon rearing, the MWMT should remain from 14 to 17°C, and proposed 17°C as the MWMT threshold. Sullivan et al. (2000) suggested that the problem with the review approach used by the ODEQ (1995) and the WDOE (2000) is that it is not quantitative, and no clear decision-making process was included for selecting appropriate thresholds.

In 1997, the California Department of Forestry and Fire Protection (CDF) made timber harvest recommendations based on a consideration of coho salmon habitat requirements (CDF 1997). They conducted a review of the available literature, and proposed an MWAT of 17.0°C, citing a personal communication with the EPA. It appears that all of the review-based approach relies almost entirely on professional judgment in lieu of methods to determine biologically defensible thresholds that will protect salmon from acute or sublethal temperature effects.

2.2.4 Risk-based approach

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Sullivan et al. (2000) conducted an in-depth review of temperature indices and thresholds used in the Pacific northwest for protecting salmonid streams. The North Coast Regional Water Quality Control Board (NCRWQCB) cited the proposed standards of Sullivan et al. (2000) in their evaluation of the Ten Mile River. They recommended listing the Ten Mile River on the 303(d) list for temperature (CRWQCB 2001). Because of the increasing acceptance of the Sullivan et al. (2000) report, and because we are unaware of any formal peer review of this report, we review it in more detail here.

Thresholds

Sullivan et al. (2000) propose using a risk-based approach for setting temperature thresholds. The authors maintain that safety factors in current temperature thresholds are based more on professional judgment than objective science. They propose a risk assessment technique, where adverse effects of temperatures are placed in an exposure context to identify population risk.

Sullivan et al. (2000) concluded that growth of juveniles during the summer would be reduced by no more than 10% of optimum if recommended upper MWMT thresholds of 16.5°C and 20.5°C for coho salmon and steelhead, respectively, were adopted. Growth modeling formed the basic approach used by Sullivan et al. (2000) to develop conclusions, and so we explored the growth modeling described in Section 5 of the paper in more detail (Appendix A).

In general, it appears that the approach taken by Sullivan et al. (2000) was scientifically based, and is clearly at the cutting edge in the development of biologically-based thresholds. However, there are some uncertainties in the way the MWAT approach was applied, particularly if it is going to be applied elsewhere. The authors' main conclusion is that the *approach* they developed is meaningful, and they do not suggest that the standards they propose be adopted generally (as some are suggesting).

The bioenergetics approach used by Sullivan et al. (2000) has many advantages. Namely, their method allows an evaluation of the effects on growth of variable temperatures that occur in a stream environment over the summer, and includes a variable for field-based consumption rates. Temperature differences among streams can be compared with a minimal amount of biological data. One problem with the approach is that it assumes that if fish are within 10% of optimum growth they are achieving biologically relevant goals (i.e., suitable end-of-summer size). Establishing standards based on allowing a 10% growth loss is arbitrary, and little data are offered to support this assumption. Sullivan et al. (2000) examined relationships between the size of juveniles in the fall and over winter survival that were developed by Holtby and Scrivener (1989) and Quinn and Peterson (1996) and concluded that a 10% reduction in summer growth would lead to a 9% reduction in over winter survival. The authors admit that this extends the relationships beyond the original data or their application, and acknowledge the uncertainty in analysis. In addition, there is no analysis addressing the potential impact of a 9% reduction in over winter survival on a population, and thus no means to determine whether this represents a biologically meaningful threshold. Although the approach was scientifically based, the selection of 10% reduction in growth as a management target requires further examination.

Sullivan et al. (2000) readily note that food supply may have more influence on population productivity than temperature. Their approach was to use a bioenergetics model to calculate consumption rates from known growth rates based on field studies in Washington streams. Sullivan et al. (2000) acknowledge that this extrapolation was a weak component of their approach, which could be particularly problematic when applying their model to other regions. If food availability is higher in northern California streams than in Washington streams, the temperature threshold they cite could be lower than is required given the food availability in northern California streams. Sullivan et al. (2000) note that site-specific consumption rates will always improve model performance. They suggest using independent measurements of food availability.

To identify a temperature regime that could be applied for the entire summer period, Sullivan et al. (2000) used the MWAT index, mostly because of its wide use. Sullivan et al. (2000) were comfortable using a short-term measure to characterize long-term temperature patterns because their data showed that short-term measures were correlated with long-term patterns. This correlation may not exist for temperature data elsewhere. In addition, using the warmest week of the summer to establish growth thresholds for the entire summer may not be biologically relevant. Good growing conditions early in the summer, for example, may compensate for poor growing conditions during peak summer temperatures, as noted by Sullivan et al. (2000).

The use of MWAT for temperature standards was first questioned in 1977 (Hokanson et al. 1977, as cited in McCullough et al. 2001). The central drawback to using MWAT is that it uses short-term exposure experiments as the basis for determining long-term temperature thresholds. Hokanson et al. (1977, as cited in McCullough et al. 2001) demonstrated that if a 19°C MWAT standard was adopted for steelhead, a typical population would experience drastic reductions from maximum net yield. They calculated that lowering the MWAT to 17°C would ensure that maximum yield would not drop more than 27% under a normal fluctuating temperature regime. McCullough et al. (2001) concluded, based on the work of Hokanson et al. (1977), that MWAT indices are not sufficiently conservative for protecting salmonid populations, and are inadequate for use as long-term temperature standards.

2.3 Exceedence

The analysis of exceedence– the amount of time a particular site exceeds a particular threshold– has been suggested as a way to account for the problems with short-term approaches. The number of days any given temperature is exceeded is far more likely to have implications for salmonid growth than the highest MWAT recorded, which may only last a short period of the year. Short-term exposure to high temperatures may not ultimately affect juvenile growth, whereas long-term exposure may.

In addition to analyzing MWAT thresholds, Hines and Ambrose (unpublished) found that the duration that streams exceeded various temperatures was highly significant in a model to predict juvenile coho presence. Exceedence appears far likelier to have implications for juvenile coho growth than the MWMT index. An MWMT, which only occurs for one week of the year will not necessarily have any influence on growth rates. In fact, researchers concerned with stream productivity often report a stream's cumulative degree-days (the summation of daily mean temperatures) because it may better indicate overall stream condition (Ward 1992). The advantage to the exceedence threshold approach is that it could begin to account for long-term temperature exposures that are likely to have the greatest impact on fish condition. There are not, however, any standards that have been developed using an exceedence-based index.

2.4 Summary

Indices such as MWAT that use the maximum weekly average of daily mean temperatures fail to account for diurnal fluctuations in temperature. Fish may respond to daily variations in stream temperatures, and using an average temperature may fail to take the impacts of daily highs and lows into account. Similarly, using the maximum weekly average of daily maximum temperatures (MWMT) may also lack biological relevance, because fish that are exposed to the daily maximum temperature for a short time might be able to acclimate, or may be able to shift to a different habitat or area to avoid high temperatures. Therefore, short-term peaks in temperature may not have significant lethal or sublethal effects.

The indices that have been widely used for establishing temperature standards are not biologically based, have never been validated in the field (Ligon et al. 1999), and have been more often based on convenience than good science. The central flaw to most approaches is that the MWAT index, which is derived from the warmest week of the summer, says very little about the growth conditions experienced by fish over the entire summer. For example, Bisson et al. (1985) and Smith (1990, as cited in Ligon et al. 1999) have both demonstrated that when food availability is high, temperatures far exceeding MWAT thresholds can result in rapid growth, rather than mortality or sublethal effects. Similarly, during periods of low temperatures other confounding variables, such as rearing densities, can result in minimal growth rates (S. Ricker, California Fish and Game, pers. comm., 2002).

Much variability in temperature standards exists in the literature (Table 1). Even standards derived using the same approach are variable (e.g., NMFS and USFWS 1997, Brungs and Jones 1977). Although some standards have more of a scientific basis than others, selection of standards from the variety reported in the literature typically appears to have been arbitrary. Temperature standards that are known to protect listed salmonid species are still lacking for northern California. Sullivan et al. (2000) for example, suggest that their approach was specific to Washington streams, and would need to be validated in other regions before being applied there.

The only two sources for standards based on temperature data specific to northern California are Welsh et al. (2001) and Hines and Ambrose (unpublished report). Both of these sources may be applicable for determining upper thermal tolerances in northern California to protect juvenile salmonids from acute mortality. However, both sources have important uncertainties, and neither addresses setting sublethal thresholds that will promote adequate growth during the summer.

3 Temperature and Forest Management Interactions

The impacts of timber harvest practices on stream temperature have been well documented since the 1960s (e.g., Beschta et al. 1987) and are described in detail in Appendix B. In general, removal of riparian vegetation results in elevated stream temperatures, which are then transmitted down stream through the channel network. Shading supplied by topography and especially by riparian canopy during the summer months is considered by many authors (e.g., Beschta et al. 1987, Brown 1969, Bartholow 2000, Rutherford et al.1997, Johnson and Jones 2000) to be essential for moderating excessive stream temperatures, particularly for small, low-order streams that are highly sensitive to changes in the thermal environment. The relatively low capacity of these streams for holding heat predisposes them to undergoing substantial temperature changes if the amount of insolation recieved is modified. Shade effects, and thermal signatures, are discussed in more detail in Appendix B.

Cumulative stream heating effects are generally considered to be represent a significant adverse impact (Bartholow 2000, Beschta et al. 1997, Beschta and Taylor 1988, Rowe and Taylor 1994, Pool et al. 2001). Because heat is not readily dissipated from a stream, and elevated temperatures can be transmitted downstream. When streams flow from exposed to shaded reaches during the daytime, exposure to direct solar radiation is reduced; however, the diffuse radiation input component still exceeds net longwave losses, and is likely to be greater than convective, conductive, and evaporative heat losses. Water temperatures may therefore be largely unaffected when passing through shaded reaches, unless groundwater seepage rates are sizable. Cumulative temperature effects and equilibrium temperatures are discussed in more detail in Appendix B.

Strategies for stream temperature prediction
Numerous modeling strategies available for exploring the impacts of timber harvest practices on stream temperatures. These strategies can be broadly categorized into statistical and stochastic schemes, and deterministic approaches. A further subdivision can be made into reach-based schemes, and basin-wide schemes. The advantages and disadvantages to these schemes are described in more detail in Appendix B.

4 Discussion and Recommendations

Many watersheds in northern California are listed, or proposed for listing, for temperature impairment under Section 303(d) by the North Coast Regional Water Quality Control Board (the Board). The temperature objective of the Board states (CRWQCB 2001):

> "The natural receiving water temperature of intrastate waters shall not be altered unless it can be demonstrated to the satisfaction of the Regional Water Board that such alteration in temperature does not adversely affect beneficial uses. At no time or place shall the temperature of any cold water be increased by more than 5°F [2.8°C] above natural receiving water temperature. At no time or place shall the temperature of warm intrastate waters be increased by more than 5°F [2.8°C] above natural receiving water temperature."

Determining natural receiving water temperature is often problematic, since historical temperature data are rare. Therefore, to determine whether or not a stream is impaired the Board relies on data and literature that report impacts to beneficial uses, particularly the effects of increased temperatures on fish condition and survival (CRWQCB 2001). The Board apparently bases its listing recommendations on the number of sites where the MWAT threshold proposed by Sullivan et al. (2000) has been exceeded (CRWQCB 2001).

While the Sullivan et al. (2000) approach is promising, its application to northern California watersheds requires consideration of regional or site-specific conditions. A scientific review panel appointed to evaluate the effectiveness of California Forest Practice Rules for protecting salmonid habitat (Ligon et al. 1999) concluded that:

> "Until a more site-specific physiological approach is used in conjunction with a watershed analysis, determining site-specific thermal requirements and impacts on salmonids as a result of timber harvesting will remain in the realm of conjecture."

We believe that the bioenergetics approach used by Sullivan et al. (2000) is a valid method with the following advantages over other approaches: (1) it can evaluate the effects of variable temperatures on growth, (2) it can incorporate field-based consumption rates, and (3) it requires a minimal amount of biological data. Before regulatory temperature standards are developed for northern California based on Sullivan et al. (2000), however, three important questions should be addressed:

1. Which indices should be used for evaluating stream temperatures in terms of salmonid habitat?
2. What are the appropriate temperature thresholds for protecting salmonids in northern California?
3. What portion of the watershed should standards be applied to?

Indices and thresholds

Temperature standards are developed to protect fish from both acute temperature effects (e.g., mortality or emigration), and sub-lethal effects, which include temperature-related reductions in growth or other vital functions that may lead to reduced reproductive fitness or mortality at a later date.

Acute Standards. Acute temperature standards could be based on an annual maximum temperature (i.e., all daily maximum temperatures should be below the threshold), or based on an MWAT. It appears that the recently developed method used by Sullivan et al. (2000) for developing an annual maximum threshold (26°C as a daily maximum for one day) uses the best available science. The standard they suggest depends on typical diurnal temperature patterns found in their study area, and they recommended conducting site-specific investigations of temperature patterns when temperatures exceed 24°C. It may also be possible to set acute temperature standards using an MWAT or a maximum weekly maximum temperature (MWMT). This type of index would apply to situations in which temperatures remain high for a sustained period (week or more) but stay below the annual maximum temperature threshold (e.g. 24°C). Welsh et al. (2001) and Hines and Ambrose (unpublished report) related the presence or absence of juvenile salmon in northern California streams to an MWAT. Their approach, modified to ensure that juvenile fish were actually present in streams prior to the onset of high temperature and corrected for confounding variables, could be used to determine if MWAT or MWMT is an appropriate index for an acute response, and if so what the appropriate threshold would be.

Sub-lethal Standards. Determining sub-lethal temperature standards that ensure sufficient growth is more problematic. It may be possible to identify indices and thresholds for northern California by using a refined application of the Sullivan et al. (2000) approach if key uncertainties can be addressed. We recommend that the following tasks be conducted before the approach is applied in northern California:

- determine the importance of summer growth for juvenile coho salmon in northern California,
- identify biologically relevant objectives for setting temperature standards to protect rearing juvenile salmonids,
- assess the appropriateness of using the MWAT index to characterize sublethal long-term temperature patterns in northern California, and
- develop realistic food availability parameters for use in bioenergetic modeling.

Importance of summer growth

One problem with the Sullivan et al. (2000) approach is that it assumes that if fish are within 10% of optimum growth during the warmest week of the year, they are achieving a biologically relevant goal. Further exploration of biologically meaningful targets for summer growth through a collaborative effort between the Water Board and other experts would be extremely valuable for determining if growth is an appropriate target, and if so what level of growth rate reduction would be acceptable, or for identifying end-of-summer size thresholds that would favor over winter survival, and ultimately marine survival.

Many uncertainties remain with regard to identifying appropriate targets for juvenile salmonid growth rates. Substrate embeddedness by fine-sediment, interspecific interactions, density-dependent dynamics, and other variables could reduce growth rates independent of temperature, making a size target less likely to be attained. There are many other variables other than size that may affect over winter survival, including availability of suitable habitat. In addition, the importance of summer growth for juvenile coho in northern California streams has not been determined. It is generally assumed that summer growth is crucial for juvenile salmonids, and this formed the basis for the thresholds suggested in the Sullivan et al. (2000) approach. In some California streams it has been observed that significant growth of juvenile salmonids occurs in the spring and fall (Shapalov and Taft 1954, Bell 2001, B. Harvey, pers. comm., 2000), and that summer growth rates can be extremely low or negative (Harvey and Nakamoto 1996, S. Ricker, pers. comm., 2002). In streams with high rearing densities, summer growth can be extremely low, regardless of stream temperatures (Harvey and Nakamoto 1996, S. Ricker, pers. comm., 2002). High summer growth rates have also been observed in streams with high summer temperatures and low rearing densities (S.Ricker, pers. comm., 2002). If summer is not an important growth period for juvenile coho in northern California, applying Sullivan et al.'s (2000) approach, which is based entirely on summer growth, would be misguided. There is enough uncertainty about the importance of summer growth for juvenile coho that standards based on growth objectives are not appropriate without (1) conducting studies to determine typical summer growth rates northern California are, and (2) determining the impacts of low growth rates on the viability of populations. If growth is verified to be a relevant objective, then the appropriateness of the MWAT index for charactering long-term temperature patterns that influence growth should be addressed.

The use of MWAT to characterize sublethal long-term temperature patterns

To develop temperature standards that could apply to the entire summer period, Sullivan et al. (2000) opted to use an MWAT index, mostly because of its widespread use. They concluded that MWAT was related to summer growth of juvenile coho salmon in Washington streams, and thus was an appropriate index. High summer water temperatures in California are likely to be longer in duration than those in Washington streams, and are likely to have different biological impacts. We recommend that summer temperature characteristics be compared between northern California and Washington streams, particularly for undisturbed streams, before the Sullivan et al. (2000) approach is applied to northern California. In addition, before MWAT is adopted as a regulatory standard, relationships between MWAT and sublethal long-term temperature patterns should be examined for northern California.

Food availability

It has long been recognized that growth rates of fish are based on the relationship between available ration and temperature regime (Brett et al. 1969). Determining ration levels in natural streams, however, is difficult. Sullivan et al. (2000) readily note that food supply has more influence on population productivity than temperature. They also state that site-specific consumption rates will always improve model performance, and suggest using independent measurements of food availability in their model. Food supply varies considerably among sites (Walters and Post 1993, as cited in Sullivan et al. 2000) and is highly unlikely to be the same in Washington streams as in northern California streams. We recommend that food availability be linked to basin-specific characteristics when applying the Sullivan et al. (2000) approach.

Rather than assuming fish have access to 30% of maximum rations in all streams, as was done by Sullivan et al. (2000), the relationships between channel type and food availability could be developed for northern California streams. Food availability is dependent on stream channel characteristics such as gradient, substrate, and riparian canopy closure (Hawkins et al. 1983, Bilby and Bisson 1989, Hetrick et al. 1998, Murphy 1998, Railsback and Rose 1999; all as cited in Sullivan et al. 2000). Developing relationships between channel types and food availability would likely entail focused field studies on macroinvertebrate populations associated with different substrate types, and different channel gradients, in undisturbed northern California stream systems. The objective would be to develop a predictive model for food availability in specific channel types. For example, it may be that gravel-bedded reaches with gradients of 2 to 4% typically produce 90% of maximum rations for fish in that reach, whereas a sand-bedded reach of 1 to 2% gradient might tend to produce only 20% of maximum rations. A GIS-based approach could be used to model food availability and energy content specific to different reaches in a basin based on their characteristics. Each reach within a basin would be assigned a particular value for food availability to be used to constrain bioenergetics models in a regional-based approach. Appropriate temperature thresholds could then be identified to achieve biological objectives given predicted food availability for each reach. One problem with this approach is the inherent variability of food production even within a pristine basin (M. Power, Seminar presented at Humboldt State University on March 11, 2002), and the influence of land management (e.g., sediment in riffles) on food availablility. Uncertainties that would need to be addressed include the relationship between invertebrate production and food availability to salmonids, and the effects of habitat characteristics and density-dependent factors (which were ignored by Sullivan et al. [2000]) on consumption rates of individual fish (Harvey and Nakamoto 1996).

Application of temperature standards

Following the development of appropriate temperature indices and thresholds, there will still be a question of where to apply the temperature standards within a basin. Even under undisturbed conditions, it is highly likely that many northern California coastal streams would have had reaches that were too hot for salmonids in the summer. If one assumes that in a forest managed for timber production that tree heights, and therefore shade, will typically be less than historical conditions, then the length of stream with temperatures unsuitable for salmon would likely increase. This issue is addressed explicitly in the EPA's South Fork Eel TMDL (EPA 1999). It is crucial to determine how much loss of suitable habitat is acceptable for maintaining and restoring coho salmon populations. We recommend that this issue be addressed collaboratively by the Board, land managers, and independent experts. The first step is to address the spatial distribution of historical high temperatures and the influence of current and future land management on temperature regimes. Other issues would then need to be explored to determine how much suitable habitat is necessary to sustain viable populations, including (1) minimum coho salmon population sizes required for viability, and (2) the distribution of habitat suitable for different coho salmon life stages (i.e., in some watersheds a small reduction in suitable reaches of stream could eliminate all the habitat for a particular life-stage).

As Welsh et al. (2001) cautioned, a standard should not be applied without considering the thermal regimes that historically occurred in those streams in the absence of management activities. Some streams may always have been too warm to support juvenile coho salmon because of natural vegetation patterns, disturbance regimes, and other environmental factors. Obviously the Water Board will not want to apply regulations that require land managers to reduce stream temperatures to below their historical potential. To develop basin-specific temperature standards, we recommend exploring the possibility of using a geographically-based approach, such as the one that was successfully applied by Stillwater Sciences to develop a temperature TMDL for the South Fork Eel River (EPA 1999).

The use of a model such as BasinTemp would allow land managers to determine historical and current temperature regimes for a basin (Figures 2, 3, 4, 5), and apply standards consistent with inherent basin potential. Further, the potential impact of a THP on the temperature regime could be estimated using the BasinTemp model, so that the overall impact of each plan within a basin could be addressed. This was successfully accomplished in the South Fork Eel River using MWAT standards, which may have their greatest application in addressing acute lethal temperature thresholds. The use of BasinTemp to develop temperature standards that ensure adequate conditions for summer growth occur, and to meet other biological objectives, would require further refinement of the model.

To determine what might be an acceptable change from reference conditions in terms of solar radiation, it may be important to address the possibility of estimating viable population sizes for coho salmon. Estimates of minimum viable population size could be used to determine the amount of habitat in a basin that could be degraded from reference conditions without threatening population viability. Temperature is only one factor that may affect population viability, but population viability estimates should still be an important consideration for developing appropriate temperature standards.

Juvenile coho rearing habitat can be mapped using attributes that are known to be correlated with habitats preferred by different life-history stages. GIS databases of channel slope, substrate, and valley confinement could be used to predict the potential suitability of specific parts of a basin for coho salmon, which could then be corroborated by field studies. When this approach is used, it becomes clear that tributaries typically provide most coho salmon rearing habitat within a river basin, and lower mainstem reaches are primarily migration corridors. If a section of stream would not have been likely to support juvenile coho salmon under undisturbed conditions, what relevance would it have to say that it is not in compliance with standards? If, for example, a temperature threshold was exceeded in a low-gradient tributary with high-quality coho rearing habitat, this would be much more of a concern to land managers than if the same threshold was exceeded in a mainstem reach. We recommend applying temperature standards to appropriate parts of the basin, based on each reach's potential ability to support juvenile coho salmon.

When developing a THP, a forester needs tools to determine the context of the plan within the basin, and how it may be affected by relevant stream temperature standards. A temperature standard that ignores spatial variability reduces the ability of land managers to tailor resource management to adequately protect juvenile salmonids. If the uncertainties described above are addressed, it may be possible to develop a method to develop basin-specific standards for regulating actions affecting water temperatures in northern California. A standard could be applied to a particular stream based on an acceptable change from historical levels of potential shade for the basin. The acceptable change could in turn be based on the reach's potential suitability as coho rearing habitat, and a temperature regime suitable for meeting biological objectives, that does not exceed acute thermal thresholds. Each THP could fit into a basin context with a specific, scientifically-based, temperature standard, and the potential effects of each THP on the temperature regime of the basin could be addressed before the plan was implemented. In portions of a basin that historically provided important rearing habitat, a THP could be structured so as to preserve all existing riparian canopy and to promote increases in riparian canopy to meet biological objectives. More management flexibility would be possible in parts of the basin that were not historically important rearing areas, or those that were not historically well-shaded.

Conclusion

Based on this analysis and review, we recommend that several issues be addressed in the short term to develop interim temperature water quality standards protective of fish in northern California. To increase the applicability of the Hines and Ambrose (unpublished report) and Welsh et al. (2001) approach, a short, focused field study that controls for stream size and habitat characteristics, could be used to assess the degree to which application of a MWAT threshold can protect juvenile coho salmon from temperatures that cause direct mortality or emigration. To address some of the concerns in the Sullivan et al. (2000) approach, available data could be analyzed to address the relationship between MWAT and long-term, sub-lethal temperature patterns in northern California, and to compare temperature characteristics in Washington vs. northern California streams. In addition, it is crucial to determine the bioenergetic ecology of juvenile coho in northern California, including the seasonal variation in food availability and seasonal growth patterns. The spatial application of temperature standards within a basin should be addressed using a modeling approach combined with a scientific analysis of the degree to which stream shading can be reduced from historical conditions while still fully protecting coho populations.

In the longer term, it would be appropriate to use a collaborative approach including the Board and other experts to address the following issues to ensure biologically relevant protections for fish in northern California:

- How important is summer growth for juvenile coho salmon in northern California?
- Is there a biological basis for determining acceptable reductions from optimum summer growth?
- Do larger end-of-summer fork-length thresholds favor increased juvenile over-winter survival, and ultimately marine survival, of juvenile coho salmon in northern California?
- How do density-dependent factors affect the influence of temperature on growth rates?
- What is the relationship between size of juvenile coho salmon during the fall and over winter survival in northern California in streams with varying levels of winter habitat quality?
- Can an over-winter survival threshold be identified that would ensure long-term health and viability of a population?
- Is it possible to examine the scales or otoliths of returning adults and determine the size that coho salmon attained prior to smolting?
- Can reach-specific food availability be predicted based on channel characteristics?
- What is the inherent variability of food production within a basin?
- How do habitat characteristics and density-dependent factors affect consumption rates of individual fish?
- Is the MWAT index related to summer growth of juvenile coho salmon in northern California streams?
- What deviation from reference conditions is acceptable for managed forestlands?
- How can minimum viable coho population size be determined for a watershed?
- Where within a basin should temperature standards be applied?

Some of these questions may be readily answered with available data. It is also possible that an interim regulatory approach based on Sullivan et al. (2000) could be implemented in northern California by addressing only its most crucial uncertainties. Any interim approach should also consider the spatial context of applying standards in a basin.

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Figures

Appendix A. Analysis of Sullivan et al. (2000)

Growth modeling forms the basic approach used in development of conclusions by Sullivan et al. (2000), and so we have explored the growth modeling described in Section 5 of the paper in some depth. We were initially concerned with the implications of the specific growth curves for steelhead presented in Figure 5.4 (p. 5-13) of the text. These curves did not seem consistent with either the general literature or with common sense. It did not seem reasonable that steelhead would require substantially cooler temperatures than coho or chinook, or that growth rates would be optimized at *lower* temperatures at high ration levels than at low ration levels. In particular, we were puzzled by the apparent inconsistencies between these implications of the plotted curves and those of the data from Wurtsbaugh and Davis (1977) on which they were based. We have concluded that the analyses were correctly conducted, and that they have been used in this paper appropriately. However, some qualitative features of the model output represent artifacts of the functional forms to which the data have been fitted, rather than properties intrinsic to the data, and could be misunderstood if taken out of context. Specifically:

1. The model may underestimate growth at high temperatures and high rations.
2. The model may underestimate the growth-maximizing temperature at high rations.
3. The model may exaggerate the sensitivity of growth rate to temperature in the neighborhood of the growth-maximizing temperature.

The model specific growth curves for steelhead, presented in Figure 5.4 (p. 5-13) of the text are reproduced here as Figure A-1. The data from Wurtsbaugh and Davis (1977) used in the derivation of these curves are superimposed, and labeled with ration levels calculated from these data using the formulae for maximum consumption rate given in the text (pp. 5-7 to 5-10). Several points are worth noting.

First, there are quite a few points for which calculated feeding rations are well over 100% of the "maximum consumption rate" parameter used in the modeling (parameter "CA" of the text). This suggests that there may be a problem with the choice CA=0.16 g/g/d used in the text.

The authors themselves note that their basic reference for growth modeling (Hanson et al. 1997) recommends values between 0.15 and 0.35 g/g/d. Considering that the data on which they based this choice contain two values greater than 0.16 g/g/d, the assertion that "the laboratory studies of juvenile fish did not support CA values greater than we selected " (p. 5-7) appears to conflict with their previous statements that "[g]enerally, the consumption at 1 gram of weight and optimum temperature is the highest consumption likely to be observed for the species" (p. 5-6) and these studies "were not designed specifically to determine the allometric relationship of consumption to weight" (p. 5-6).

Second, the temperatures at which growth is maximized are in all cases less than 14°C. This is substantially less than the growth-maximizing temperature of 17°C found for coho, even though one might expect steelhead to be more temperature tolerant than coho on the basis of their ecology. Moreover, the modeled growth-maximizing temperature for steelhead is *lower* at 100% rations than 80% rations, a behavior that is not easy to explain on physiological grounds. We believe that this is an artifact of the functional forms chosen for the modeling. Because most of the data represent ration levels much smaller than 100%, the behavior of the model at high rations depends more on extrapolation from low- to mid-ration data than on the actual high-ration data.

A non-parametric treatment of the data, using multivariate kernel smoothing[1], is shown in Figure A-2. Ration levels found in wild populations are typically well below 100% (cf. Figure 5.8 of the

[1] Analysis performed in S-Plus, using the routine `sm.regression` of the "sm" library (Bowman and Azzalini 1997), with smoothing parameters `h=(2,0.2)`.

text, p. 5-21), and it is harder for fish to maintain a given feeding ration as temperatures increase (because the metabolic demand and maximum consumption rates rise, while food ability remains relatively fixed). Therefore the behavior of the model at the combination of high rations and high temperatures is not necessarily very important in the context of setting temperature criteria for fish. One would expect the model to be more reliable at moderate feeding rations, and indeed, the applications of the model to wild populations in the text (pp. 5-14 to 5-30) confirm that this is the case.

However, there is another aspect of the Sullivan et al. (2000) model that might have some consequences for setting temperature criteria. The criteria reported in the text are based on a risk analysis, where acceptable risk is defined as a 10% growth loss. The temperature interval corresponding to this at a given ration level will depend on the "flatness" of the corresponding growth curve at its peak. However this "flatness" is strongly influenced by choice of functional forms in the model. Comparing Figures A-1 and A-2, the empirical data seem to be consistent with more "flatness" than the model is capable of representing.

The growth model is actually a composition of two submodels: one for growth rate as a function of temperature, consumption rate, and initial fish weight, and another for consumption rate as a function of temperature. In the text the first of these is assumed to be linear combination of T, T^2, C, C^2, and CT, where T is the temperature and C is the consumption rate, fitted to the Wurtsbaugh-Davis data. As shown in Figure A-3, this model fits quite well, just as the authors assert. However, the fact that the model is quadratic in temperature necessarily limits the "flatness" it is capable of exhibiting. Non-quadratic variants of this model, using T^8 or 1/(24-T) in place of T^2, give rise to equally good fits.

The second submodel, the model for consumption rate as a function of temperature, is based on a Thornton-Lessem function. However, because this function is awkward to work with in Excel, the authors approximate this with a cubic polynomial. The relationship between these two curves is shown in Figure A-4. The cubic is indeed a good approximation, as the authors assert, and certainly adequate for the purposes of the paper. However, the approximating cubic is substantially less flat near its maximum than the original Thornton-Lessem curve, and this "loss of flatness" propagates to the final growth curves.

Whether this is a problem is not clear. We could presumably push the temperature criteria up or down a little bit by repeating the risk analysis with alternate model forms, but in our opinion such an exercise is unlikely to produce significantly different results. We think it is more useful to simply note that the model used in this paper, like all models, relies on some simplifying assumptions, which might limit its range of applicability. In particular, the criteria in this paper might require revisiting for a population known or suspected to feed at rates corresponding to high ration levels.

Appendix B. Timberland/Temperature Management Interactions

The impacts of timber harvest practices on stream temperature have been well characterized since the 1960's (e.g., Beschta et al. 1987). Paired basin studies and pre- and post-harvest assessments have identified profound changes in stream temperatures, especially during the summer months, when clear-sky conditions coupled with low-flow discharge combine to expose streams lacking any appreciable canopy cover to intense heating. Routinely, these elevated stream temperatures are transmitted down through the channel network. Streamside vegetation also contributes to several other ecosystem functions, such as bank stability, channel roughness, and woody debris.

Vegetation removal can have a variety of affects both locally and basin-wide. These include changes in air temperature, increased variability of wind patterns in the vicinity of the stream, reductions in relative humidity, a likely increase in surface albedo, and increased soil temperatures and hence interflow temperatures. Stream channel changes could include changes in channel downstream, width-depth adjustments, channel aggradation, bank instability, and so forth.

Numerous studies have addressed stream temperature response to timber harvesting. Table B.1 and B.2 summarize some of these studies for East Coast (and Japan) and Pacific Northwest examples. Data are shown only for summertime maximum temperatures, but reveal post-harvesting and clear-cutting changes of up to 11.2°C. Diurnal fluctuations (not shown) reveal even bigger changes as the loss in canopy cover prevents any nighttime trapping of longwave radiation, resulting in considerable stream cooling.

Recovery of stream temperatures to pre-harvest levels may take many years. Beschta (1989) suggests that stream temperature recovery in the Oregon coast ranges after clearcutting requires a decade, but might require approximately two decades in the Cascades region.

Even after reforestation, stream temperatures may not fully recover. Hewlett and Fortson (1982) found that after several years of regeneration of a clearcut in the Piedmont area of the SE USA, the bufferstrip was inadequate to control summertime temperatures. They suggested that the gently rolling relief permitted more solar radiation exposure and penetration into the soil column, resulting in higher interflow temperatures. St-Hilaire et al. (2000) observed that direct solar radiation in the soil column is increased in clear-cut areas and that this increase modifies both the thermal and hydrologic budgets. They incorporated soil characteristics into a deterministic heat budget model (CEQUEAU) and used it to simulate stream temperatures for a stream in New Brunswick. They concluded that in spite of the prevalence of surface heat budget terms, locally advected water through interflow, might be an important contributor.

Table B.1. Summer stream temperature response to harvesting. East Coast and Japan examples[a]

Location	*Treatment*	*Temperature metric*	*Change (°C)*	*Reference*
Georgia	Clearcut with partial buffer	Av. Jun-Jul Max	+6.7	Hewlett and Fortson (1982)
Maryland	Riparian harvest	Av. summer Max.	+4.4 – 7.6	Corbett and Spencer (1975)
New Jersey	Riparian herbicide	Av. Summer Max.	+3.3	Corbett and Heilman (1975)
North Carolina	Deadened cove vegeation	Av. Summer Max.	+2.2 – 2.8	Swift and Messer (1971)
	Complete clearcut	Av. Summer Max.	+2.8 – 3.3	Swift and Messer (1971)
	Understory cut	Av. Summer Max.	0 – 0.3	
Pennsylvania	Riparian harvest	Av. Summer Max	+3.9	
	Clearcut with herbicide	Av. Jun-Jul Max.	+10 – 10.5	Rishel et al. (1982)
	Commercial clearcut with buffer strip	Av. Jun-July Max.	+0.6 – 1.6	Rishel et al. (1982)
West Virginia	Clearcut	Av. Summer Max.	+4.4	Kochenderfer and Aubertin (1975)
Virginia	Riparian vegetation removal.	Av. Jul Max.	+1 – 3.0	Pluhowski (1972)
Japan	Clearcut	Daily Max.	+4.0	Nakamura and Dokai (1989)
New Hampshire	Clearcut	Daily Av. For hottest month	+4.0	Likens et al. (1970)
West Virginia	95% clearcut with thin buffer	Av. Weekly for growing season	+1.7	Aubertin and Patric (1974)

[a]adapted from Beschta et al. (1987) and Binkley and Brown (1994)

Table B.2. Stream temperature response to harvesting. Pacific Northwest examples[b].

Location	*Treatment*	*Temperature metric*	*Change (°C)*	*Reference*
Alaska	Clearcut, natural openings	Δ temperature per 100m	+0.1 – 1.1 / 100m	Meehan (1970)
British Columbia	Logged	Av. Jun-Aug diurnal range	+0.5 – 1.8	Holtby and Newcombe (1982)
	Logged and burned	Av. Jun-Aug diurnal range	+0.7 – 3.2	Holtby and Newcombe (1982)
Oregon (Cascades)	Clearcut	Av. Jun-Aug Max.	+4.4 – 6.7	Levno and Rothacher (1967)
	Clearcut and burned	Av. Jun-Aug Max.	+6.7 – 7.8	Levno and Rothacher (1967)
Oregon (coast range)	Clearcut	Av. Jul-Sep Max.	+2.8 – 7.8	Brown and Krygier (1967)
	Clearcut and burned	Av. Jul-Aug Max.	+9-10	Brown and Krygier (1970)
Oregon (Cascades)	Mixed clearcut	Δ Temperature per 100m	+0 – 0.7 / 100m	Brown et al. (1971)
	Tractor striped	Δ Temperature per 100m	+15.8 / 100m	Brown et al. (1971)
Oregon	25% clearcut, thin buffer	Av. Daily for hottest 3-weeks	+2.5 – 3.0	Harr and Fredericksen (1988)

British Columbia	66% clearcut, no buffer	Av. Daily for July	+5.0	Feller (1981)
Alaska	Clearcut	Annual Maximum	+10.0	Moring (1975)
	Clearcut	Maximum diurnal Flux	+11.2	Moring (1975)
Oregon (Cascades)	Paired study, Clearcut vs undisturbed	Mean Weekly Max.	+5.4 – 6.4	Johnson and Jones (2000)
Oregon (Cascades)	Clearcut and burned	Ave daily Max. for hottest 10-days	+6	Beschta and Taylor (1988)
Oregon	Burned	Maximum change	+3.3 – 19	Amaranthus et al.(1989)
Northern California	Logged and 'roaded'	Maximum change	+3.3 – 9.4	Kopperdahl (1971)

[b]adapted from Beschta et al. (1987) and Binkley and Brown (1994).

Mechanisms of stream heating

The mechanisms responsible for stream heating are well understood, and comprehensive reviews on the subject can be found (e.g., TVA 1972). Figure B-1 shows the principle fluxes that contribute to stream temperature heating. As water flows downstream, the rate of heat gain or loss is the sum of these fluxes, primarily net radiation, evaporation and condensation, convection, conduction, and advection. Biochemical fluxes and heat energy released by frictional effects are not shown in Figure B-2. These fluxes contribute only a small portion of the total energy budget for a stream, especially during summer months. Heat released by frictional energy dissipation is the smallest flux and is routinely omitted from energy budget calculations (e.g., Raphael 1962, Vugts 1974), although Webb and Zhang (1997) show that in select instances (steep headwater channels during winter) frictional heat release can predominate. Theurer et al.(1984) include a friction flux term in the SNTEMP (Stream Network Temperature Model),which contributes toward estimation of maximum stream temperatures.

During summertime clear-sky conditions predominate in mid-latitude regions. During this season, the net radiation energy budget is dominated by solar radiation, chiefly direct solar radiation, which contributes up to 80% of the total short wave radiation flux (wavelengths 0.3μm to 4.0μm) (Monteith and Unsworth 1990). Comprehensive discussions of solar radiation can be found in TVA (1972), Iqbal (1983), Ice (2001), and Adams (2001). Shading supplied by topography and especially by riparian canopy during the summer months is considered by most authors (e.g., Brown 1969, Beschta et al. 1987, Rutherford et al. 1997, Bartholow 2000, Johnson and Jones, 2000) to be essential for moderating excessive stream temperatures; particularly for small, low order streams, which are highly sensitive to changes in the thermal environment. The relatively low heat carrying-capacity of these streams predisposes them to significant temperature changes if the amount of insolation receit is drastically modified.

Shade effects

Brown (1983) estimated that daytime net radiation beneath a fully canopy may be 15% or less than that of an unshaded stream, while Pluhowski (1972) showed that net radiation was 6 times greater at an unshaded site than at a proximal site where trees effectively shaded the stream from the sun. Ice (2001) reports three studies showing the importance of shade. Moore at el. (1999, as cited in Ice 2001) performed an experiment with four tanks, two deep and two shallow. One of each was shaded with plywood, but with air exchange permitted. The authors reported that diurnal heating and cooling increased with decreasing depth in the tank, and that air temperature had relatively little influence on the rate of heating or cooling. Ice (2001) also reports on studies that artificially shaded an irrigation depth to varying degrees, and noted a complete reduction in maximum stream temperature with 100% shading compared to an upstream reach. Ice (2001) reports on a study that applied a shade cloth to a recent debris flow runout in the H.J. Andrews watershed in Oregon. Temperatures were reduced by up to 1-3°C at the bottom of the shaded 200-meter reach.

Other mechanisms

Of the other fluxes, streambed conduction is considered important in select cases, especially when the streambed is composed of bedrock or coarse gravel and/or cobble (Brown 1969, Comer and Grenney 1977). Several researchers have incorporated bed conduction fluxes in stream temperature prediction models (e.g., Evans et al. 1998, Sinokrat and Stefan 1993, 1994). Crittendon (1978) showed that stream temperature for a small, low-gradient stream with no groundwater inputs, was controlled mostly by wind speed and bed conduction. As Bartholow (1989) observes, however, these two parameters were varied across 1-2 orders of magnitude in the sensitivity analyses.

Statistically, daily maximum air temperatures are strongly correlated with stream temperature, providing a rationale for the development of statistical relationships between air and water temperature (e.g., Smith 1981, Mitchell 1999, Stefan and Preud'homme, 1993). However, this doesn't imply any causal mechanisms between high air temperature and water temperature. Indeed, if warm air is present over a stream surface in the absence of strong wind currents, heat gain by convection will tend to offset the evaporative heat losses.

Bartholow (1989) performed a sensitivity analysis of the USFW SNTEMP model and found that stream temperature was very sensitive to air temperature and moderately sensitive to percent shade, relatively humidity, velocity, upstream temperature, and stream width. He observes though, that, "[I]n terms of heat flux, atmospheric radiation dominates most of the time, especially in summer" (Bartholow 1989, p.12).

The thermal signature debate

While there is general agreement about which mechanisms comprise stream energy budgets, there is considerable debate over which factors predominate. Bartholow (2000) identified the dichotomy between those who consider that solar radiation is critical in controlling stream temperature (making the protection of riparian vegetation essential (Beschta et al. 1987, Beschta 1997), and those who consider that a warm environment and ambient air temperature control stream heating (Sullivan et al. 1990, Larson and Larson 1996, 1997, 2001, Zwieniecki and Newton 1999). The latter view is not well supported with data. Larson and Larson (1996) significantly underestimated the major contribution of direct solar radiation in summertime stream heating, and chose a poorly shaded region of stream to study; their data thus underestimate the effects of shade provided by continuous riparian cover in the majority of cases. Beschta (1997) observed, for the Upper Grande Ronde in northeastern Oregon, that during the summertime 90% of the fish-bearing streams in subwatersheds have wetted widths of 10 feet or less. Shade effects on narrow streams are likely to be much greater than on the relatively wide stream studied by Larson and Larson. Zwienieki and Newton (1999) advocate the notion of stream thermal signatures, and suggest that elevated temperatures as a result of clear-cutting return to 'normal' trends in downstream recovery zones. Their work hinges on quantifying natural warming trend relationships using field measured stream temperatures in upstream 'clearcut' and downstream 'recovery' zones. However the sample sizes (a maximum n=7, and in two cases, n=6) are too small to define robust relationships. Furthermore, the choice of quadratic relationships in the curve-fitting exercise for the high-discharge cases is considered highly tenuous. In short, the authors fail to provide convincing arguments against downstream temperature transmittance.

The debate extends into the arena of cumulative effects, with the same authors (Sullivan et al. 1990, Caldwell et al. 1991, Zwieniecki and Newton 1999) suggesting that local effects of clear-cutting or other disturbances on heat fluxes are rapidly ameliorated in downstream "recovery zones". Burton and Likens (1973) showed rapid recovery of temperatures in streams that flowed from clearcut to fully forested reaches in the Hubbard Brook Experimental Forest, although the authors could provide no explanation for this recovery. Caldwell et al. (1991) concluded that downstream cumulative effects were not evident for their study of Type 4 streams flowing into Type 3 streams in Washington State under a variety of vegetation conditions. Their sample size however (n=9), is very small. Furthermore, measurements of shade for the upstream and downstream reaches was made using canopy densiometers which have been shown to provide unreliable estimates of streamside riparian shading (Cook et al. 1995, Ice 2001).

By contrast, several authors stress the importance of cumulative stream heating effects (Rowe and Taylor 1994, Bartholow 2000, Beschta et al. 1997, Beschta and Taylor 1988). Because heat is not readily dissipated from the stream, elevated temperatures upstream can be transmitted downstream, elevating those downstream regimes. When streams flow from exposed to shaded reaches during the daytime, there is a reduction in direct solar radiation, but the diffuse component is still greater than net longwave losses, and likely to be greater than convective, conductive, and evaporative heat losses. Hence, water temperatures will be largely unaffected when passing through shaded reaches, unless groundwater seepage rates are sizable.

Increasing discharge can certainly moderate stream temperatures by diluting warm temperatures with cooler water. Several authors (Brown 1969, Hockey et al. 1982, Gu 1998) have identified the important role of discharge in stream temperature heating. The Brown equation (1969, and described in more detail below) recognizes that stream heating is proportional to the surface area of the stream receiving atmospheric radiation, and inversely proportional to discharge.

An indication of the importance of low-order (1^{st} and 2^{nd} order) stream channels in the cumulative effects debate is provided below with low-flow volume data that have been calculated (using field-measured (Stillwater Sciences, unpublished data collected in the South Fork Eel River basin, California and North Umpqua River basin, Oregon in 1997 and 1998) low-flow hydraulic geometry relations and making the simplifying assumption that cross-sectional channel geometry is everywhere rectangular) for sub-basins in the Oregon Cascades region and in sub-basins in the South Fork Eel basin in Northern California (Tables B.3 to B.7). The stream channel networks for the Oregon sub-basins are comprised of 1:24,000 USGS blueline hydrography with an automated channel extraction routine that defines a channel wherever drainage area equals or exceeds 10 hectares. The result is a channel network that better captures the full drainage density in low-order valleys and hollows. The subbasins in the South Fork Eel basin are exclusively comprised of 1:24,000 USGS blueline hydrography. The latter likely underestimates perennial channel drainage densities, and the former probably over estimates drainage density. Nevertheless, the striking features of these data are the large volumes (approximately 25-45%) of water contained in 1^{st} and 2^{nd} order channels. These are the most sensitive portions to temperature of the stream channel network, and are typically characterized by shallow depths. Consequently, these low thermal inertia streams respond rapidly to changes in thermal regimes, and can be important in their aggregate downstream transport of elevated stream temperatures.

Table B.3. Stream Order versus Low-flow Volume and Channel Length. Steamboat Creek, North Umpqua, Oregon.

Stream Order	Low-flow Volume (m^3)[a]	Stream length (km)
1	123572 (26.3%)	604
2	56593 (12%)	201
3	57060 (12%)	117
4	63088 (13.4%)	61
5	27781 (6%)	13
6	142660 (30.3%)	27

[a]Percentages have been rounded and hence may not sum to 100

Table B.4. Stream Order versus Low-flow Volume and Channel Length. Rock Creek, North Umpqua, Oregon.

Stream Order	Low-flow Volume (m^3)	Stream length (km)
1	69157 (29.2%)	360
2	39196 (16.6%)	138
3	30826 (13.05%)	66
4	25907 (11.0%)	33.5
5	25868 (11.0%)	15.6
6	45250 (19.2%)	13.5

[a]Percentages have been rounded and hence may not sum to 100

Table B.5. Stream Order versus Low-flow Volume and Channel Length. Canton Creek, North Umpqua, Oregon.

Stream Order	Low-flow Volume (m^3)	Stream length (km)
1	38012 (26.4%)	211
2	22763 (15.8%)	83.8
3	21081 (15.6%)	48
4	10182 (7.1%)	15.5
5	10918 (7.6%)	9
6	40994 (28%)	16.5

[a]Percentages have been rounded and hence may not sum to 100

Table B.6. Stream Order versus Low-flow Volume and Channel Length. Bull Creek, South Fork Eel Basin, Northern California.

Stream Order	Low-flow Volume (m^3)	Stream length (km)
1	7606 (19.5%)	90
2	5745 (14.7%)	28.6
3	1248 (3%)	3
4	24393 (62.6%)	18.7

[a]Percentages have been rounded and hence may not sum to 100

Table B.7. Stream Order versus Low-flow Volume and Channel Length. Rattlesnake Creek, South Fork Eel Basin, Northern California.

Stream Order	Low-flow Volume (m^3)	Stream length (km)
1	4011 (13.7%)	93.8
2	3213 (11%)	20
3	7167 (24.5%)	15.5
4	14918 (51%)	11.8

[a]Percentages have been rounded and hence may not sum to 100

Table B.8. Stream Order versus Low-flow Volume and Channel Length. Elder Creek, South Fork Eel Basin, Northern California.

Stream Order	Low-flow Volume (m^3)	Stream length (km)
1	716 (21%)	16
2	529 (15.5%)	3.9
3	2173 (63.5%)	4.4

[a]Percentages have been rounded and hence may not sum to 100

Equilibrium Temperature

The theoretical concept of equilibrium temperatures and maximum equilibrium temperatures has been invoked by several authors (Sullivan et al. 1990, Larson and Larson 1996, 1997, Zwienieki and Newton 1999), who argue that the local ambient environment is preeminent in determining stream temperatures, and that the local changes in the thermal regime are not transmitted downstream.

Edinger et al. (1968) explored the concept of the equilibrium temperature with regard to the temporal and spatial distribution of temperature within water bodies. While not explicitly calculating any equilibrium temperatures, they found the concept to be a useful tool for identifying the role of meteorological factors on water temperature.

The equilibrium temperature of a water surface is the temperature at which the net exchange of energy with the atmosphere is zero. If only the conduction and convection fluxes were involved, then the equilibrium temperature would equal the air temperature near the surface. However, other heat exchange processes are relevant (primarily solar radiation and air vapor pressure), and thus equilibrium temperature cannot be linked exclusively to air temperature.

The equilibrium temperature is an elusive quantity that cannot be measured (Rutherford et al. 1993) but must be predicted by heuristic methods. If radiation, air temperature, humidity, and wind speed could be held constant, river temperature would eventually approach the equilibrium temperature and the net heat exchange would approach zero. In practice, these fluxes vary significantly throughout the day.

The notion that an equilibrium temperature can be calculated (or even measured) and then related to water temperature is a tenuous one at best. Even if the quantity can be calculated, the assumptions that water temperature is driven toward an equilibrium temperature and at a rate driven by the difference between the two, and that meteorological variables are chiefly responsible, ignore the fact that the rate of absorbed radiation (the H_r term in Edinger et al.'s equation 18) terms are embedded in equilibrium equations. Focusing exclusively on the meteorological variables and ignoring any alterations in the stream environment as a consequence of shade reduction and thus elevated solar radiation receipt, ignores the prime driver of summertime stream heating.

As Edinger et al. (1968) state, "[B]ecause T_e [the equilibrium temperature] and K [the thermal exchange coefficient] are defined in terms of meteorological variables only, they may be employed without reference to historical records of water temperature" (p.1142). And indeed, nowhere do they actually calculate an equilibrium temperature. To do so would require measurements of all the fluxes necessary to calculate stream temperature. Those who have sought to apply the equilibrium concept to local stream heating relations have ignored this fact.

The concept remains a useful mathematical tool that has application for exploring how an unaltered reach might respond to elevated heat loading from upstream.

Strategies for stream temperature prediction

Toward the objective of exploring the impact of timber harvest practices on stream temperatures, there are numerous modeling strategies available. These can be broadly categorized into statistical and stochastic schemes, and deterministic strategies. A further subdivision can be made into reach-based schemes, and basin-wide schemes

Statistical and stochastic schemes

Statistical schemes are dominated by regression analyses, typically correlating air temperature with stream temperature (e.g., Smith 1981, Stefan and Preud'homme 1993, Mitchell 1999). These schemes take advantage of the strong statistical relationship between water temperature and various air temperature metrics, and also the fact that air temperature is comparatively widely measured metric as opposed to stream temperature. Examples of stochastic schemes include Chui and Isu's (1978) application of Kalman filtering to temperature modeling.

Advantages:

1. Simple to calculate.
2. Minimal input data requirements.

Disadvantages:

1. Only applicable for the range of data used to compute the regression.
2. Non-transferable outside of the area where the data were collected.
3. Doesn't allow scenario testing.
4. No insight into causal mechanisms.

Deterministic schemes

Simple deterministic schemes such as the Brown equation (Brown 1969, 1970, 1980) have been widely applied to assess stream temperature response to harvesting activities (e.g., Cafferata 1990). The Brown equation takes on the following form:

$$T_w = (\Sigma H \times A / Q) * 0.000267$$

where T_w = predicted temperature change (°F)
ΣH = rate of heat absorbed by the stream (Btu/ft^2 min^{-1})
A = surface area (ft^2)
Q = discharge (cfs)
0.000267 = constant converting discharge from cfs to lbs/minute.

Advantages and disadvantages of the Brown equations, which are largely shared amongst all simple deterministic schemes are:

Advantages:

1. Simple to apply.
2. Few input data requirements.
3. Transferable.

Disadvantages:

1. Only applicable for relatively short reaches (< 500 m).
2. Requires reach-averaged shade measurements, ignoring spatial variability of shade, and thus predictions perform less well in fully forested reaches.
3. Doesn't allow scenario testing of various shade levels along a reach.
4. Doesn't allow for identification of causal mechanisms if the primary assumption (the amount of direct radiation striking the stream surface) doesn't hold.

Fully deterministic, spatially distributed modeling schemes purport to capture the full physics of stream heating, providing predictions at high temporal (sub-hourly, daily, weekly, and so forth), and fine spatial (reach and segment scale) resolution. Several authors (Raphael 1962, Morse 1970, Rutherford et al. 1993, 1997, St-Hilaire et al. 2000) have developed stream temperature prediction models using well-described energy budget principles (TVA 1972). The Stream Network temperature model (SNTEMP, Theurer et al. 1984) has been widely used to predict longitudinal temperature caused by alterations in flow regimes, dam release, and impacts of canopy modification (Theurer et al. 1984, Bartholow 1989, Mattax and Quigley 1989). The model is a steady-state 1-D heat transport model that predicts daily mean and maximum temperatures.

The Hydrological Simulation Program-Fortran (HSPF, Bicknell et al. 1997) has similarly been applied in a variety of circumstances. Chen at al. (1998a, 1998b) developed a more elaborate shading module (SHADE) and heat flux model for streambed conduction, and incorporated them within HSPF to explore the interaction between riparian forest management and stream temperature. The revised model was tested in the Upper Grande Ronde basin in Oregon. The differences between observed versus model predictions were 2.5°C and 2.2°C for 1991 and 2.1°C and 2.1°C for 1992 for summer maximum and 7-day average maximum temperatures. More egregious however, were significant temporal offsets between model predictions and observed values. The study highlighted the importance of collecting reliable, local estimates of shading relations. In their study, shade parameters were averaged over 1000-meter segments, and thus ignored important finer scale resolution.

Advantages:

1. Capture full mechanisms of stream heating.
2. High temporal and spatial resolution.
3. Allows full scenario testing of pre/post impact analysis.
4. Because they're based on first principles, these models are readily transferable to a variety of different basins.

Disadvantages

1. Expensive to operate.
2. Considerable input (meteorological and hydrological) data requirements.

3. Require considerable parameter adjustment for those variables (wind speed, evaporation) that can vary widely over short distances and short timeframes.

Mention should also be made of recent technological advances in airborne remote sensing techniques, which have great potential for temperature modeling endeavors. Thermal infrared (TIR) imaging has been available for some time, but has become more widely applied to water temperature assessments in the Pacific Northwest recently (Torgersen et al. 1999, 2001, Kay et al. 2001). The technique uses thermal imagers mounted on helicopter platforms to acquire emitted thermal infrared radiation (8-14μm) from the upper 0.1mm surface of streams. The imagers acquire data at very high resolutions (0.2 – 0.4 meters) for the entire channel. Radiant water temperatures correspond to kinetic water temperatures for a range of stream environments within ±0.5°C. While the technique holds much promise for the near future, it's not yet at operational mode. The cost and the need to resolve several limitations (assumptions of complete vertical mixing, problems of atmospheric thermal radiation interference, spatial-temporal resolution issues, and so forth), means that it remains primarily a research tool. However, its utility in providing calibration data for conventional stream temperature models, and its superior ability to identify local thermal refugia means it does have a potentially important role.

Reach-based model sensitivity

Toward developing the ability to explore local and downstream cumulative effects of clearcutting on stream temperature, Bartholow (2000) parameterized the SSTEMP (Stream Segment Temperature model) with realistic values for a Pacific Northwest example, and explored the downstream effects from upstream elevated stream heating. SSTEMP was developed as a subset of SNTEMP and is applicable for single time-step, single reach examples. The model applies the same equations as SNTEMP, but was designed as a Microsoft Windows application. Bartholow found clearcutting increased mean daily temperatures by 2.4°C and maximum temperatures by 3.6°C. Stream shading was the most sensitive variable, accounting for 1.48°C of the increase in daily maximum, with stream width the second most sensitive variable, accounting for 1.35°C.

We elaborated upon Bartholow's study by showing a hypothetical example of reach-based cumulative effects for rather more extreme conditions than modeled by Bartholow. In his study, he simulated the impact of a fully forested and an *extensively* clearcut reach on downstream temperatures. We compared a near fully-forested reach (85% shade – values based on EPA effective shade estimates for South Fork Eel TMDL,EPA 1999) and a *fully* clearcut East-West orientated reach. We largely applied the same parameters that Bartholow used in his study, although we did make changes to the latitude, time-of-year, relative humidity, and ground temperature values. Emphasis is placed on mean temperature predictions because maximum (and by extension, minimum) predictions are less reliable (Bartholow 2000, Sullivan et al. 1990). Model predictions are shown in Table B.9, revealing a +4°C difference in mean water temperature between the shaded and unshaded reach (Table B.10). We took the analysis one step further, and used the predicted mean temperature for the forested and clearcut examples as input to three different downstream scenarios (Table B.11). The results show water temperatures flowing from clearcut to downstream clearcut get progressively warmer. More striking is the prediction for mean water temperature flowing from clearcut down through the forested reach. That temperature remains +4°C higher than if that stream had flowed exclusively through forested to forested reaches.

Table B.9. SSTEMP parameters for a hypothetical E-W orientated stream.

Parameter	*Value*
Latitude (degrees)	40.0
Date	July-31
Downstream Elevation (m)	1000
Channel Length (km)	10.0
Channel slope (m/m)	0.0305
Width's A Term (s/m²)	4
B Term where $W = A*Q^B$	0
Manning's n	0.035
Air Temperature (°C)	25
Relative Humidity (%)	25
Wind Speed (mps)	0.186
Ground Temperature (°C)	15
Thermal gradient (j/m²/s/C)	1.65
Possible Sun (%)	100
Dust Coefficient	5
Ground Reflectivity (%)	15
Solar Radiation (j/m²/s)	345.355
Total Shade (%)	0% or 85%
Segment Azimuth (degrees)	90

Table B.10. Comparison of temperature predictions for unshaded versus near-complete shading (85%).

	0% shade	**85% shade**
Predicted mean temperature (°C)	16.5	12.4
Estimated maximum temperature (°C)	29.8	15.5
Estimated minimum temperature (°C)	3.2	9.4

Table B.11. Comparison of temperature predictions for water flowing from an upstream clearcut zone to different downstream riparian conditions.

	0% to 0% shade condition	**0% to 85% shade condition**	**85% to 85% shade condition**
Predicted mean temperature (°C)	20.37	17.32	13.19
Estimated maximum temperature (°C)	31.27	19.68	16.58
Estimated minimum temperature (°C)	9.47	14.96	11.22

A further analysis (B-2) showed mean temperature predictions for different vegetation heights for the East-West Orientated stream and for different stream widths. The analysis revealed the role of riparian shading in reducing stream temperatures for streams of intermediate width (5-15

meters), but showed that for wider streams, the effectiveness of stream shading is increasingly diminished. Narrow streams (< 5 meters) have reduced surface area, and thus (in this model example) are less dependent on riparian shading to moderate mean temperatures.
While only a hypothetical example with extreme parameter values for shading, the exercise demonstrates the utility of reach-based modeling schemes to explore cumulative effects, using realistic parameterizations.

Basin-Scale Modeling Strategies
For basin-wide stream temperature assessments, modeling schemes, which account for the spatially varying thermal regime throughout the channel network are required. Depending on resources available, a variety of different schemes may be adopted, ranging from the fully-mechanistic schemes, to the more statistically orientated strategies. An additional approach takes advantage of the ability of Geographic Information Systems to organize and stratify spatial information and couple it with numerical temperature prediction models. LaMarche (1998) coupled a simple solar radiation model (SolarFlux, see review in Dubayah and Rich 1995) with a numerical heat balance model to explore stream heating in two reaches of the Deschutes River in southwestern Washington State. Another example of this hybrid approach is provided by the BasinTemp (EPA 1999) stream temperature model. The model is a quasi-mechanistic, basin-scale model that predicts reach-based (where reaches are defined as approximately 25-30 meters in length) stream temperatures.

BasinTemp Model Assumptions
The model assumes that direct solar radiation is the chief mechanism responsible for stream heating during clear-sky, summer low-flow conditions in mid-latitudes. Topographic and, especially, riparian shading, are the chief controls over of the amount of direct solar radiation receipt at the stream surface. The model draws on existing research on solar radiation and stream temperature physics to develop a simple model structure that uses a minimum of field-measured stream temperature data for calibration but does not require comprehensive field data for model parameterization. The model predicts a wide variety of temperature metrics, including MWATs, and weekly and monthly averages. Its steady-state scheme and simplifying model assumptions precludes predicting water temperature at more finely resolved timesteps (e.g., daily maximum and minimums). Fully mechanistic schemes are required to predict these diurnal temperature fluctuations.

The model was originally developed to support a temperature TMDL for the South Fork Eel river basin in North California (http://www.epa.gov/region09/water/tmdl/proposed.html). One of the main features of the model is its ability to explore different management scenarios by explicitly adjusting reach-based riparian vegetation parameters and assessing the resulting impacts on predicted stream temperatures.

The model couples a GIS-based solar radiation model (the Image Processing Workbench, Frew 1990, Dozier and Frew 1990, Dubayah et al. 1990, Dubayah 1995) with a 1-D steady-state heat balance model. Digital elevation data and stream channel data (Figure B-3) are combined with vegetation information (Figure B-4), the latter which may be acquired from remotely sensed data sources, aerial photography, or ground-based surveys. For the examples shown here, vegetation data was acquired from Landsat TM sources. Tree height information for individual vegetation assemblages was calculated using various diameter at breast height (DBH) to height relationships.

The radiation model calculates spatially varying, daily averaged radiation predictions (comprising all the components of atmospheric radiative flux) for individual reaches for a given date and given latitude (Figures B-5, B-6). These data provide the principle input to the steady-state 1-D heat balance model. The important role of riparian shading is revealed for two sub-basins within the Bull Creek watershed in Northern California. Squaw Creek (Figure B-5) is characterized by old growth redwood and hence reveals comparatively little change in solar radiation receipt from current vegetation conditions to a 'reference' vegetation condition. The reference vegetation condition assumes tree height in the riparian zone are raised to late-seral tree heights. By contrast, Cuneo Creek (Figure B-6), is low-gradient tributary to Bull Creek that has experienced considerable near-stream disturbance and which is well illustrated by the elevated amounts of solar radiation reaching the stream surface. The reference vegetation scenario (Figure 2) for this stream illustrates the important role riparian shade.

The 1-D heat balance model predictions are improved by optimizing them using measured thermograph data for the basin of interest. Optimization parameters, while not having direct physical interpretation, represent aggregation of several physical mechanisms, and thus have physical relevance. The utility of the optimization scheme, which uses 4-5 optimization parameters, is that it precludes the necessity to gather considerable quantities of expensive and hard to assemble meteorological and hydrological data. Furthermore, the model scheme explicitly acknowledges the necessity for parameter tuning, as opposed to the fully-mechanistic schemes (e.g., SNTEMP) which, while purporting to model the full physics of stream heating, also have to engage in elaborate parameter tuning, often adjusting parameters to physically unrealistic values.

The energy balance model provides temperature predictions at the reach scale and then routes those temperatures downstream. This feature, coupled with the ability to supply spatially varying riparian vegetation geometries, explicitly permits the exploration of cumulative stream heating effects for a range of vegetation scenarios.

Model Application

The model has been applied to basins in the South Fork Eel River in Northern California, and several basins in the North Umpqua watershed, Oregon. Results for the Bull Creek basin, an approximately 40 square mile sub-basin within the South Fork Eel are shown in Figure B-7. Eight thermographs distributed throughout the basin were used to calibrate and optimize the model. Comparison of observed versus predicted temperatures show a very high degree of correlation ($R^2 = 0.99$, and RMSE = 0.17°C).

Model parameters for Bull Creek were used in a comparison of MWAT temperature predictions for two additional sub-basins within the South Fork Eel (Figure B-8). Elder Creek is one of last remaining old-growth Douglas fir basins in Northern California. By contract, the Rattlesnake Creek sub-watershed is characterized by gently rolling relief and low gradient channels flowing throwing scrub-dominated meadows. The results nicely illustrate the important role of vegetation in controlling summertime stream temperatures during the hottest period of the year. MWAT predictions for Rattlesnake Creek are necessarily much warmer than those for Elder Creek, where the old-growth Douglas fir provide almost complete canopy cover. The graph has useful implications for land management, suggesting that shifts in warmer (e.g., Rattlesnake-like temperature) to cooler (e.g., Elder Creek-like temperatures) can be achieved by riparian canopy adjustments. This implication was explored further for the Rattlesnake Creek example, by testing different riparian management scenarios (supplying different riparian tree heights for different stream classes), and assessing the resulting changes to the predicted MWATs. Interestingly, even the most ambitious riparian management scenario had little effect on downstream temperatures toward the mouth of Rattlesnake Creek. This example illustrates that there may be instances where a combination of local terrain characteristics and/or comparatively large channel width-to-depth ratios may preclude effective riparian management strategies.

Applications of the BasinTemp model demonstrate that simple, physically-based schemes which capture the chief mechanisms driving stream heating, can be useful for exploring reach-scale and cumulative temperature effects. Incorporating the ability to adjust local riparian vegetation (or discharge, for example) relations and assessing downstream impacts is an important feature which is relevant to identifying the impacts of timber harvest practices on stream temperature.

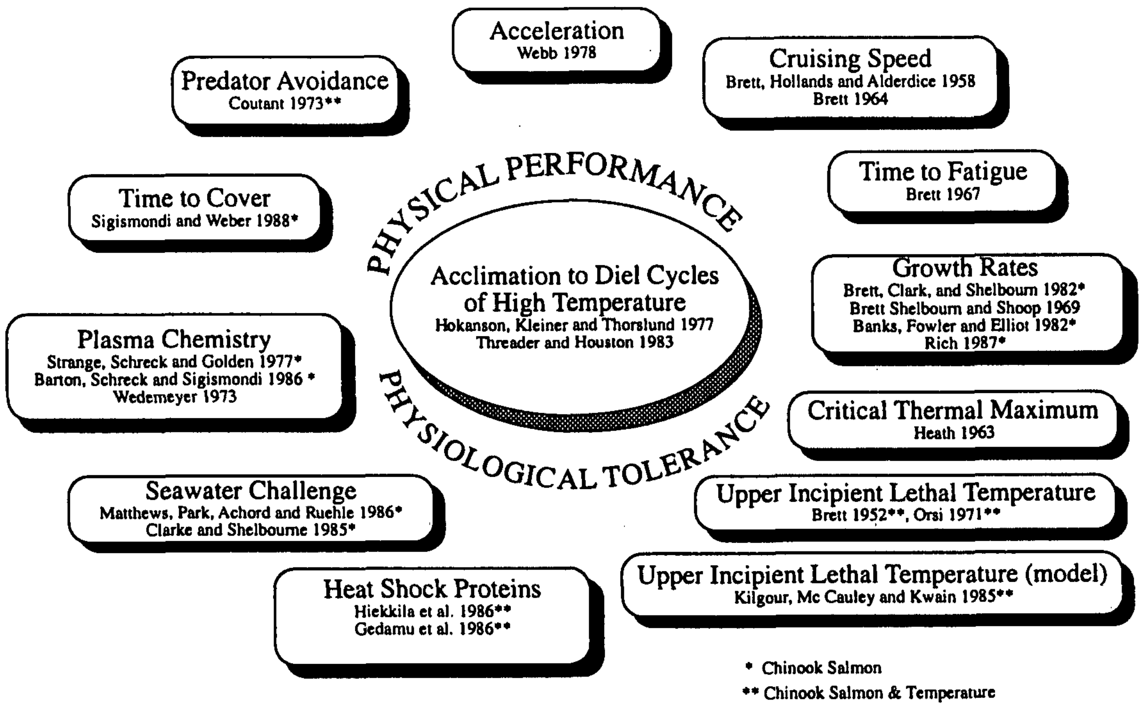

Figure 1. Approaches to evaluating the effects on fish of high temperatures and of prior exposure to high temperatures.

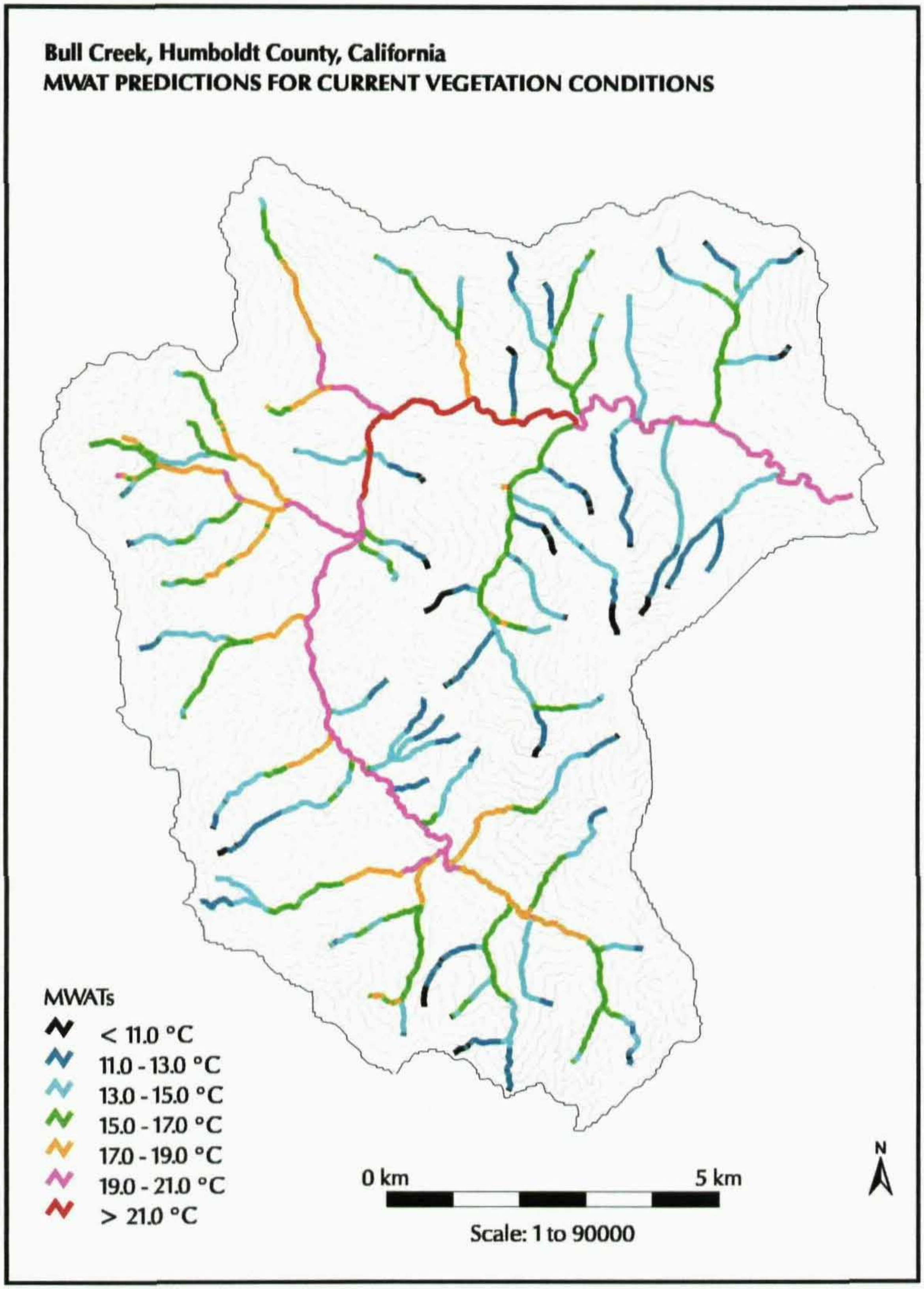

Figure 3. MWAT predictions for current vegetation conditions in Bull Creek, California.

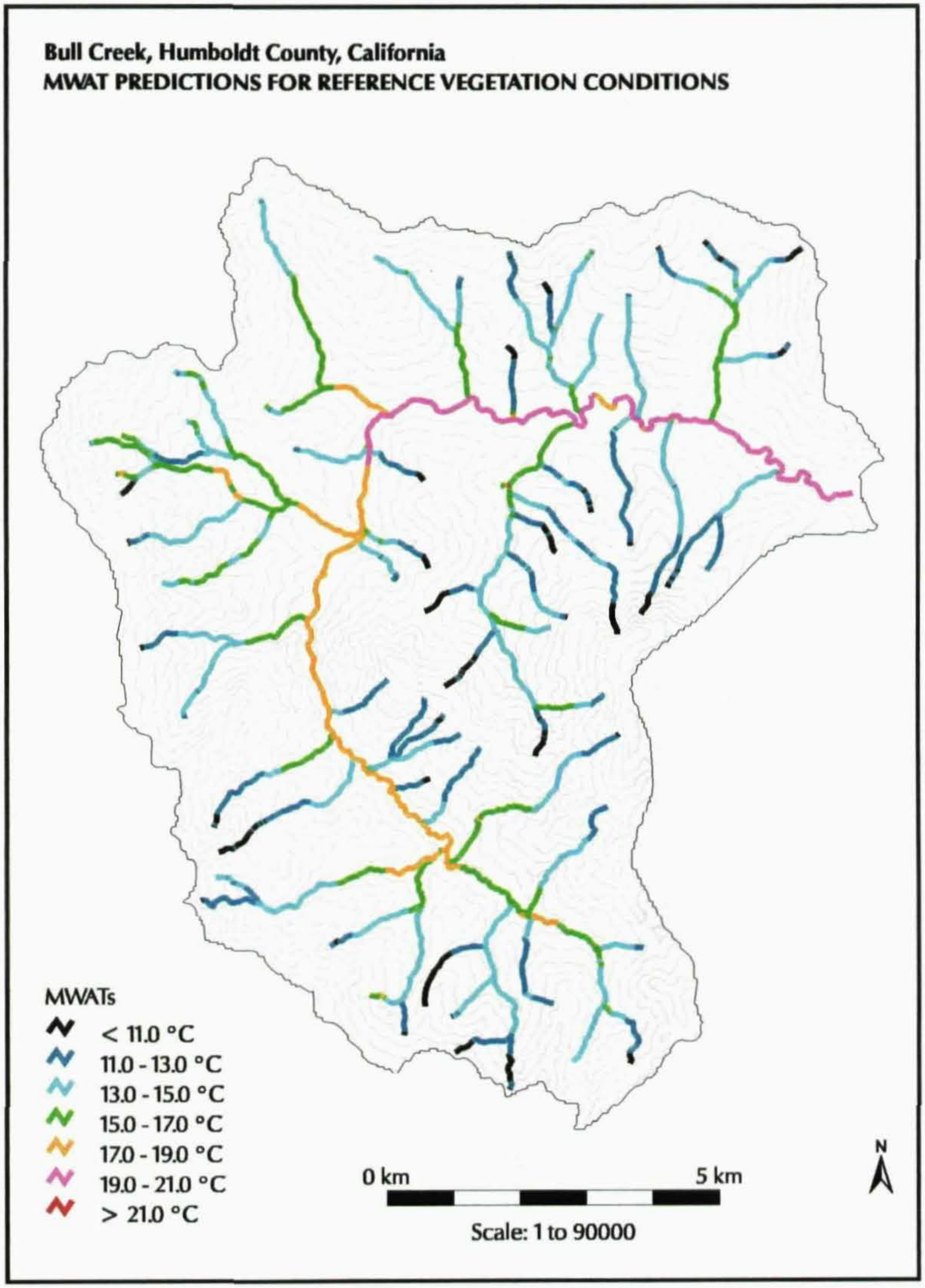

Figure 2. MWAT predictions for reference vegetation conditions in Bull Creek, California.

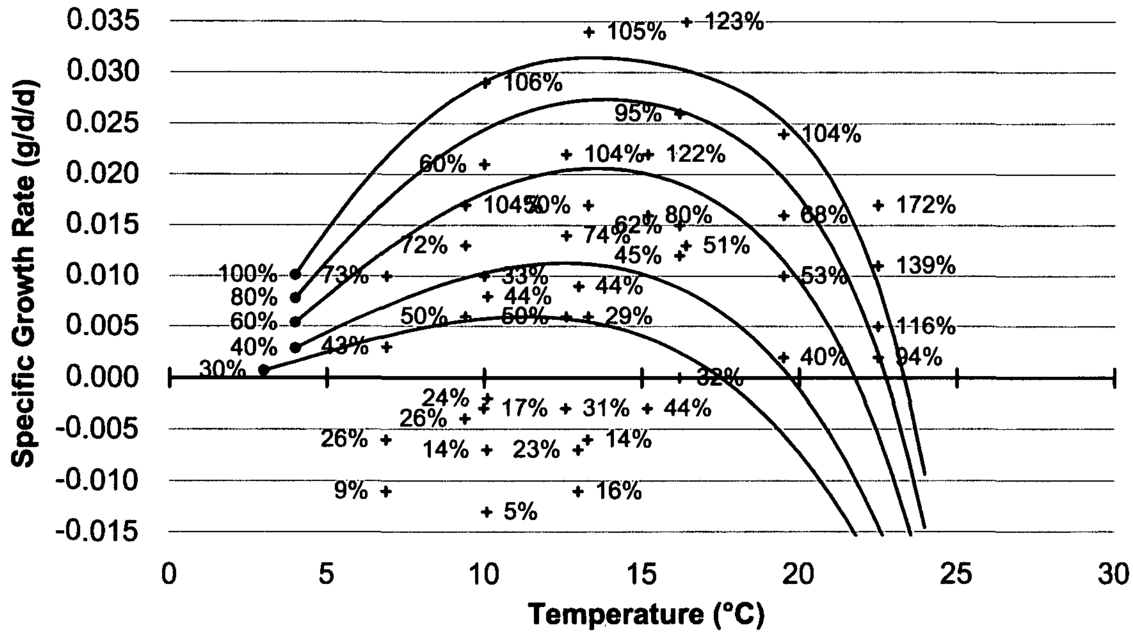

Figure A-1. Specific growth rates predicted for steelhead salmon by the model of Sullivan et al. (2000) at 30%, 40%, 60%, 80%, and 100% of *ad-libitum* feeding levels, and observed specific growth rates from Wurtsbaugh and Davis (1977).

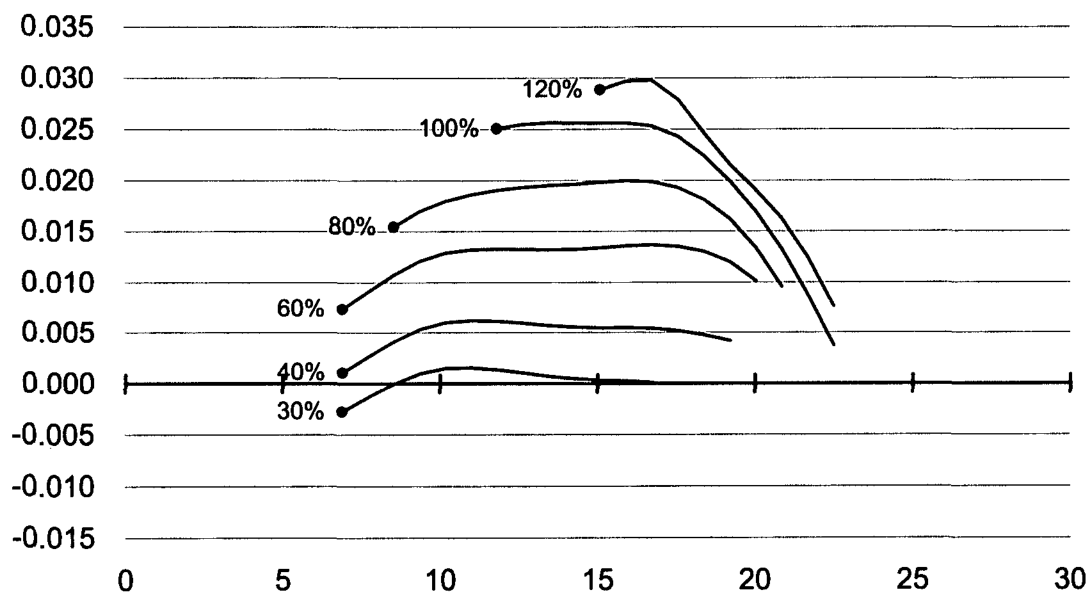

Figure A-2. Specific growth rates predicted for steelhead salmon at 30%, 40%, 60%, 80%, 100%, and 120% of *ad-libitum* feeding levels by a non-parametric model fitted to data of Wurtsbaugh and Davis (1977).

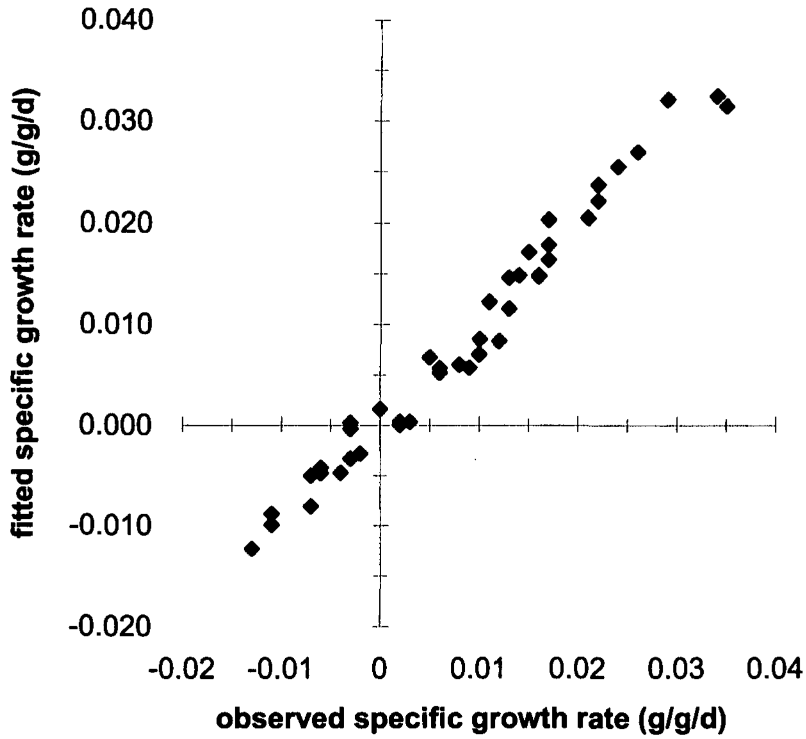

Figure A-3. Predicted and observed specific growth rates for steelhead. Quadratic model of Sullivan et al. (2000), fitted to data from Wurtsbaugh and Davis (1977).

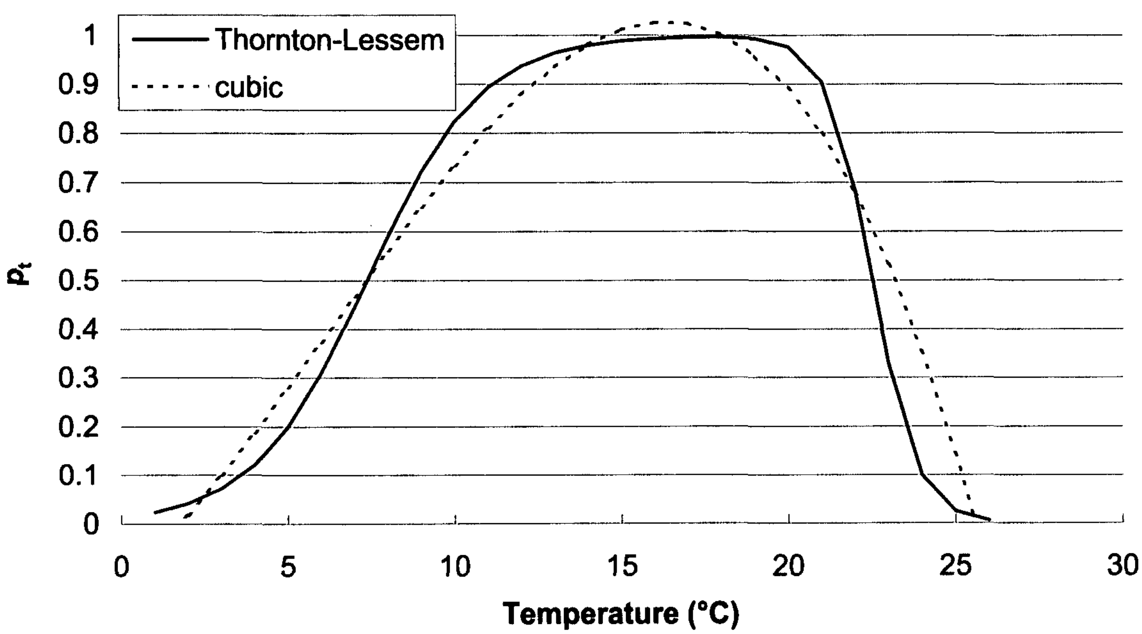

Figure A-4. Thornton-Lessem model for the dependence of consumption, expressed as a fraction of the maximum consumption at optimal temperature, and the cubic approximation to this model used in Sullivan et al. (2000).

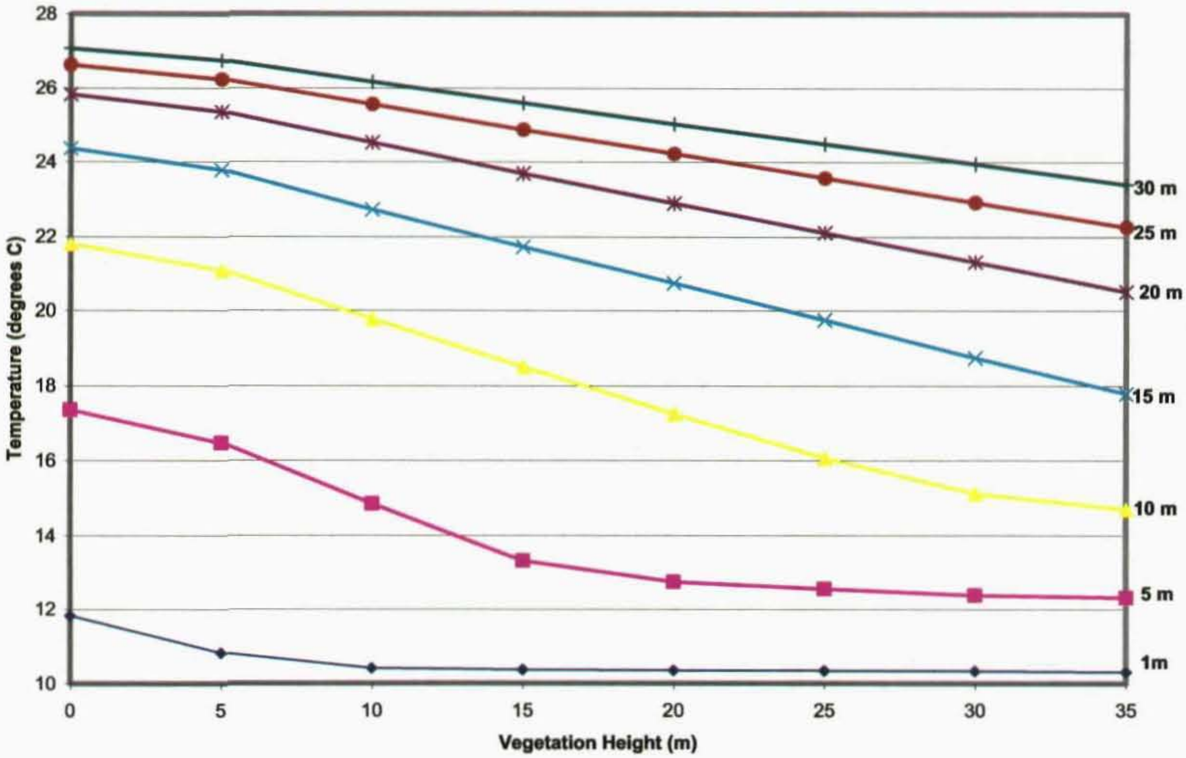

Figure B-2. SSTEMP predictions of mean temperatures for different stream widths and riparian vegetation heights.

Figure B-3. Elevation and stream channel data for Bull Creek, California.

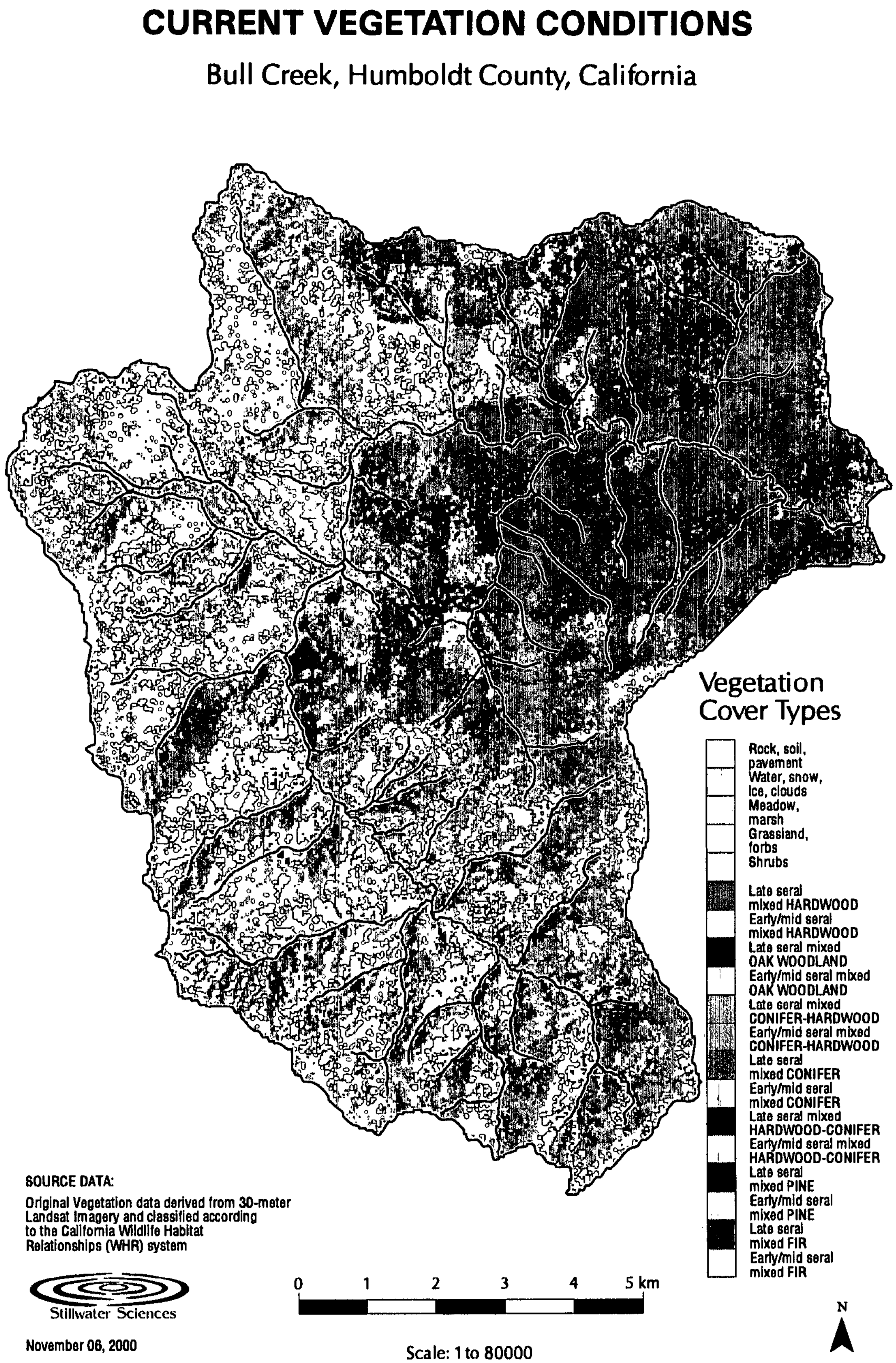

Figure B-4. Vegetation conditions in Bull Creek, California.

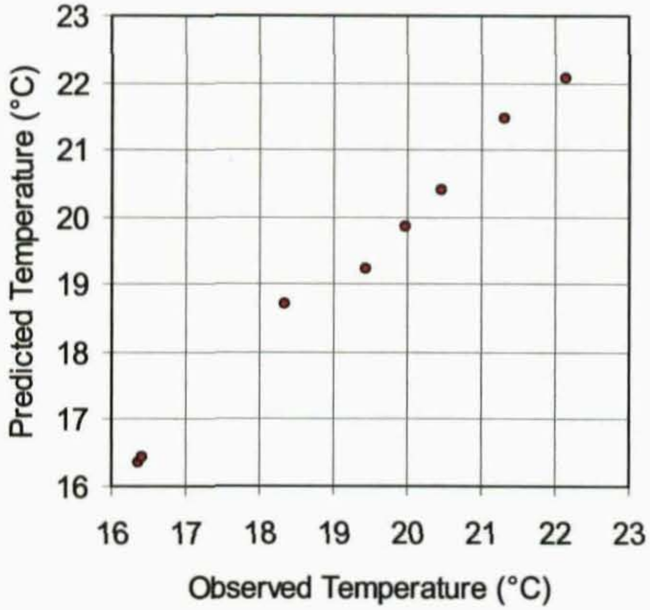

Figure B-7. Observed versus predicted MWATs for current vegetation conditions. Bull Creek, Northern California.